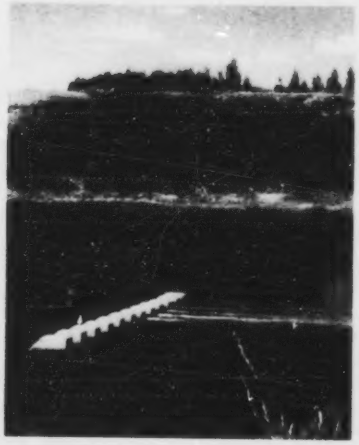
NUTRIENTS AND THEIR IMPACT ON THE CANADIAN ENVIRONMENT



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Nutrients and their Impact on the Canadian Environment Erratum

Chapter 3 - pg. 20. Figure 3.4, bar colours. Edmonton's sewage has undergone secondary treatment since 1935 (not untreated). Sewage was released 'Untreated' prior to 1910 in Toronto, 1901 in London, 1937 in Winnipeg, 1971 in Saskatoon, 1932 in Calgary, and in 1913 in Edmonton (black bar is incorrect).

Chapter 3 – pg. 38, 2nd para., last sentence should read '…: a total of 139 thousand tonnes of P and **384** (not 482) thousand tonnes of N was applied to 2 579 thousand hectares of Canadian cropland in 1996.

P

Nutrients and Their Impact on the Canadian Environment

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The report, *Nutrients and their Impact on the Canadian Environment*, provides information on sources of nutrients (nitrogen and phosphorus) to Canadian air, water and soil, and effects of these emissions on the health of humans and ecosystems. This assessment was conducted under the direction of an interdepartmental working group of officials from Environment Canada, Fisheries and Oceans Canada, Agriculture and Agri-Food Canada, Natural Resources Canada, and Health Canada. Working group members provided overall direction on the approach, as well as technical information on specific topics. Members of the working group were:

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Executive Summary

In 1994, the Parliamentary Standing Committee on Environment and Sustainable Development reviewed the Canadian Environmental Protection Act (*CEPA*) and identified several issues related to nutrients and their environmental impact and management: first, that only one class of nutrient is currently regulated (namely phosphorus in laundry detergents) and, second, that other nutrient classes and sources may be adversely affecting the environment (HCSCESD 1995). In response to these concerns, the Government of Canada committed to undertake "a comprehensive study of nutrients that enter the environment through human activities ... to determine whether or not nutrients in general are causing negative environmental effects; whether only certain nutrients, rather than nutrients as a class, are problematic, and; whether those effects are limited ... to water or to entire ecosystems, including wildlife" (Government of Canada 1995). This report meets the Government of Canada's commitment to document the nature and extent to which nutrients derived from human activity may be impairing Canadian ecosystems and affecting the quality of life and health of Canadians. This assessment was undertaken by representatives from five federal departments (Environment, Natural Resources, Agriculture and Agri-Food, Fisheries and Oceans, Health) under a Memorandum of Understanding to encourage collaboration and co-ordination amongst these departments in the use of science and technology for sustainable development.

Based upon our assessment of nitrogen (N) and phosphorus (P) loading from Canadian sources and the effects of these nutrient loads on forest, freshwater and coastal ecosystems, it is clear that nutrients are causing problems in certain Canadian ecosystems and affecting quality of life for many Canadians. We are certain that N and P loading from human activity has:

- accelerated eutrophication of certain rivers, lakes and wetlands in Canada, resulting in loss of habitat, changes in biodiversity and loss of recreational potential.
- increased the frequency and spatial extent to which the drinking-water guideline for nitrate has been exceeded in ground waters across Canada and caused economic burden for those Canadians who must transport household water from off-site sources.
- caused and continues to cause fish kills due to ammonia toxicity.
- contributed to a decline in amphibians in southern Ontario due to long-term exposure to elevated nitrate concentrations.
- led to elevated risks to human and livestock health through increased frequency and spatial extent of toxic algal blooms in Canadian lakes and coastal waters.
- contributed to acidification of soils and lakes in southern Ontario and Québec and resulted in incipient N saturation in some forested watersheds.
- increased carbon production in Canada's forests due to N deposition.
- increased concentrations of the potent greenhouse gas N_2O and increased concentrations of nitrogen oxides contributing to formation of photochemical smog in certain Canadian cities.
- contributed to quality of life concerns for Canadians through water use impairments (e.g., excessive algal and aquatic weed growth; blockages of screens and filters) and aesthetic (taste and odour) concerns related to water supplies.
- increased the economic burden to Canadians as a result of the need for treatment, monitoring and remediation of contaminated water.

We estimate that more than 12 thousand tonnes of P and 304 thousand tonnes of N entered Canadian fresh, ground, and coastal waters in 1996 as a result of human activity. The largest point source was municipal sewage, adding an estimated 5.6 thousand tonnes P (as total P) and 80 thousand tonnes N (as total N). Discharge of industrial wastewater added at least 2.0 thousand tonnes P and 11.8 thousand tonnes N (as nitrate and ammonia) to Canadian surface waters. The industrial loads are an under-estimate, however, as data were not available for New Brunswick, Nova Scotia, Prince Edward Island, and, in the case of Québec, industries discharging to waters other than those in the St. Lawrence River Basin. There are no national estimates of nutrient losses due to leaching or runoff from agricultural fields. A recent assessment of N losses from agricultural land where the soils have a water surplus predicted that 17% of Ontario, 6% of Québec and 3% of Atlantic farmland would produce runoff or seepage water with > 14 mg N/L (MacDonald 2000b). In British Columbia, 5% of the agricultural land has a water surplus and 69% of this area was predicted to generate runoff or seepage water with N concentrations > 14 mg/L.

At least 1 400 thousand tonnes N were released to the atmosphere in 1996 as a result of human activity in Canada. The largest single source of N emissions to the atmosphere was agricultural activity, in particular the release of ammonia (NH_3) associated with handling and application of manure and fertilizer. Release of nitrous oxide (N_2O) was split approximately evenly among industrial, transportation-related, and agricultural sources. In the case of nitric oxide and nitrogen dioxide (i.e., NO_x), the largest emissions were from industrial and transportation-related sources; NO_x emissions are not available for agricultural emissions but are likely similar to industrial emissions (Janzen et al. 1998). Reliable data were not available for P emissions to air.

At least 4 287 thousand tonnes of N and 449 thousand tonnes of P were added to Canadian land in 1996. Of this, 1 968 thousand tonnes N and 442 thousand tonnes P were applied to cropland in Canada in the form of fertilizer, manure, and biosolids. An additional 773 thousand tonnes N were added to cropland as a result of nitrogen fixation by legumes and 43 thousand tonnes N atmospheric deposition of nitrate and ammonium. Harvesting of crops removed 386 thousand tonnes P and 2 491 thousand tonnes N, or 87% of the added P and 89% of the added N. Nutrients were also added to non-agricultural land by atmospheric deposition, N fertilization of British Columbia coastal and interior conifer forests managed for timber harvest, and landfilling of industrial and municipal waste.

In fulfillment of the objectives of this assessment as identified by the Government of Canada (1995), we conclude that:

- nutrients released to the environment from human activity are impairing the health of certain ecosystems, contributing to quality of life concerns for Canadians and, on occasion, endangering human health.
- nutrient impacts tend to be associated with certain nutrient forms. For example, most inland waters in Canada are intrinsically P limited and thus addition of P has accelerated eutrophication. In contrast, leaching of nitrate-N has increased the frequency and spatial extent to which the drinking-water guideline for nitrate has been exceeded in ground waters. Concerns about nutrients in the atmosphere relate largely to the role of dissolved inorganic N (in the form of nitrate and ammonium) as a fertilizer to forest ecosystems and a contributor to acid rain, NO in urban smog production, and N_2O in greenhouse warming.

- although the predominant and most demonstrable impacts to date have occurred in aquatic ecosystems and caused water use impairments, the first symptoms of negative effects on forest ecosystems have also been observed.

In Canada, environmental problems caused by excessive nutrients are less severe and tend to be more localized than in countries with a longer history of settlement and agricultural production. This situation is due to our relatively small population compared to our land base and the protective measures implemented by both the federal and provincial/territorial governments in the last 30 years. Given the problems already caused by nutrient loading in Canada, as well as newly-emerging concerns, it is critical that gains achieved by improved wastewater treatment and other control measures not be reversed by relaxation of standards or by failure to keep pace with population growth. The best and most advanced science should continue to be integrated into practical solutions to maintain or improve the quality of Canadian air, water and soil environments.

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1.0 Introduction

Highlights

- Nutrients are chemical substances that provide nourishment and promote growth of micro-organisms and vegetation. They include nitrogen, phosphorus, carbon, hydrogen, oxygen, potassium, sulfur, magnesium and calcium, as well as other elements required in smaller quantities: iron, zinc, copper, manganese, boron, molybdenum and chloride.
- A surplus of nutrients in an ecosystem can lead to increased plant growth and changes in biodiversity. Ammonia, nitrate, and nitrite are also toxic to aquatic and terrestrial animals, including humans, when present in quantities well in excess of requirements.
- An oversupply of nitric oxide to the atmosphere can contribute to the creation of acid rain and urban smog.
- This report meets the Government of Canada's commitment, in its response to the 1994 review of the *Canadian Environmental Protection Act* (CEPA), to determine the nature and extent to which nutrients derived from human activity may be impairing Canadian ecosystems and affecting the quality of life and health of Canadians.

1.1. Issue Context

The purpose of this report is to meet the commitment made by the Government of Canada to determine the extent to which nutrients derived from human activities may be impairing Canadian environments, contributing to quality-of-life concerns for Canadians, or endangering their health.

Nutrients are elements essential to plant growth and are present naturally in the environment; however human activity has increased the supply of biologically reactive forms of nutrients, particularly nitrogen (N) and phosphorus (P). Nutrient addition from human activity includes runoff water from land cleared for agriculture, forestry, mining, urban expansion and recreational developments; losses associated with agricultural practices such as fertilizer and manure application to soils and cultivation of crops such as legumes that fix N; industrial emissions to air, soils and water; and household wastewater discharge.

This report examines the role of nutrients, particularly N and P derived from human activity, in contributing to excessive plant production in aquatic and terrestrial ecosystems, thereby, affecting fish and wildlife; imposing a toxic threat to plants and animals, including humans; and, in contributing to quality of life concerns for Canadians. Nitrogen and phosphorus should only be referred to as "nutrients" when examined in the context of plant growth. Nutrient toxicity, specifically nitrogen toxicity, occurs when an element essential for growth is available in quantities well in excess of requirements. Nitrogen does not serve a nutritive role in phenomena such as nitrogen-induced acidification or nitrogen-oxide-enhanced ozone production. This report focuses only on the "nutritive" effects of N and P on Canadian environments.

Nutrients Defined

Nutrients are essential elements for all organisms. Nutrients include nitrogen, phosphorus, carbon, hydrogen, oxygen, potassium, sulfur, magnesium and calcium (collectively known as macronutrients because they are needed in relatively large amounts) as well as other elements required in smaller quantities (micronutrients such as iron, zinc, copper, manganese, boron, molybdenum and chloride). These elements are present naturally in air, soil and water.

The *1999 Canadian Environmental Protection Act* (CEPA) defines a nutrient as any "substance or combination of substances that, if released in any waters, provides nourishment that promotes the growth of vegetation". Under the Act, the Governor in Council may, on the recommendation of the Minister, make regulations "for the purpose of preventing or reducing the growth of aquatic vegetation that is caused by the release of nutrients in waters and that can interfere with the functioning of an ecosystem or degrade or alter, or form part of a process of degrading or altering, an ecosystem to an extent that is detrimental to its use by humans, animals or plants" (CEPA 1999 Part 7, Division 1, 118(1)).

Because the Government of Canada committed to a comprehensive study of nutrients that would determine whether effects are limited to water or to entire ecosystems including wildlife, the definition of nutrients for this assessment was expanded beyond the CEPA definition to include terrestrial ecosystems. For the purpose of this assessment "nutrients" included any element that:

- has nutritive value, in particular the two major nutrients nitrogen and phosphorus;
- may be supplied in excess as a result of human activity; and
- may interfere with the normal functioning of aquatic or terrestrial ecosystems.

Assessment of ecosystem effects also included more than the CEPA endpoint of prolific growth of aquatic vegetation.

1.2. Federal Control Programs and Actions to Date

With the identification of the critical role of nutrients (specifically P and N) in controlling eutrophication of aquatic systems, countries around the world began to implement controls for reducing nutrient inputs to aquatic ecosystems. In Canada, the federal government instituted the *Canadian Environmental Protection Act* (CEPA) in 1988, which is the only federal act to address environmental concerns related to nutrients directly.

CEPA gives the federal government authority to regulate nutrients that can cause excessive growth of algae and aquatic weeds. Phosphorus in laundry detergents is the only nutrient source currently regulated. The *Phosphorus Concentration Regulations* (DOE 1989), the only regulation made under this part of CEPA, prohibit the manufacture, use and sale of laundry detergents that contain P in concentrations greater than 2.2 % phosphorus (by weight).

The *Fertilizers Act*, administered and enforced by Canadian Food Inspection Agency, also addresses nutrients although not from the perspective of adverse environmental impacts of nutrients *per se*. The *Fertilizers Act* regulates the safety, efficacy, and labeling of fertilizers (including animal manure) and supplements imported into or sold in Canada. It identifies as nutrients: nitrogen, phosphorus and potassium (primary nutrients); calcium, magnesium, and sulfur (secondary nutrients); iron, zinc, copper, manganese, boron, molybdenum, chloride and, in certain situations, sodium (micronutrients). The fertilizer definition excludes soil supplements defined as "any substance or mixture of substances, other than a fertilizer, that is manufactured, sold, or represented for use in the improvement of the

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1.3. Rationale for this Study

In recognition of the fact that science is constantly evolving, CEPA was passed with the intent that it be reviewed by Parliament every five years. The 1994 review of CEPA identified two issues related to nutrients and their environmental impact and management: that only one class of nutrient (phosphorus in laundry detergents) is currently regulated and that other nutrient classes (e.g., nitrogen compounds) and sources (e.g., other cleaning agents, fertilizers, industrial wastewater) may be adversely affecting the environment (HCSCESD 1995).

In their review of the 1988 CEPA (HCSCESD 1995), the Parliamentary Standing Committee on Environment and Sustainable Development recommended that Environment Canada:

- "regulate the phosphate content of cleaning agents other than laundry detergents under Part III of CEPA";
- "determine whether regulation is required for nutrients in cleaning agents other than phosphates", and
- "determine whether sources of nutrients other than cleaning agents are adequately regulated by the provinces and territories, and if they are not, that the powers of Part III be broadened to include these other nutrient sources".

In response to the recommendations of the Parliamentary Standing Committee, the Government of Canada committed to undertake "a comprehensive study of nutrients that enter the environment through human activities ... to determine whether or not nutrients in general are causing negative environmental effects; whether only certain nutrients, rather than nutrients as a class, are problematic, and; whether those effects are limited ... to water or to entire ecosystems, including wildlife" (Government of Canada 1995). The Government of Canada is therefore committed to assess scientifically the nature and extent of damage to the environment caused by nutrients from sources other than laundry detergents prior to any further regulation or other control action on nutrients.

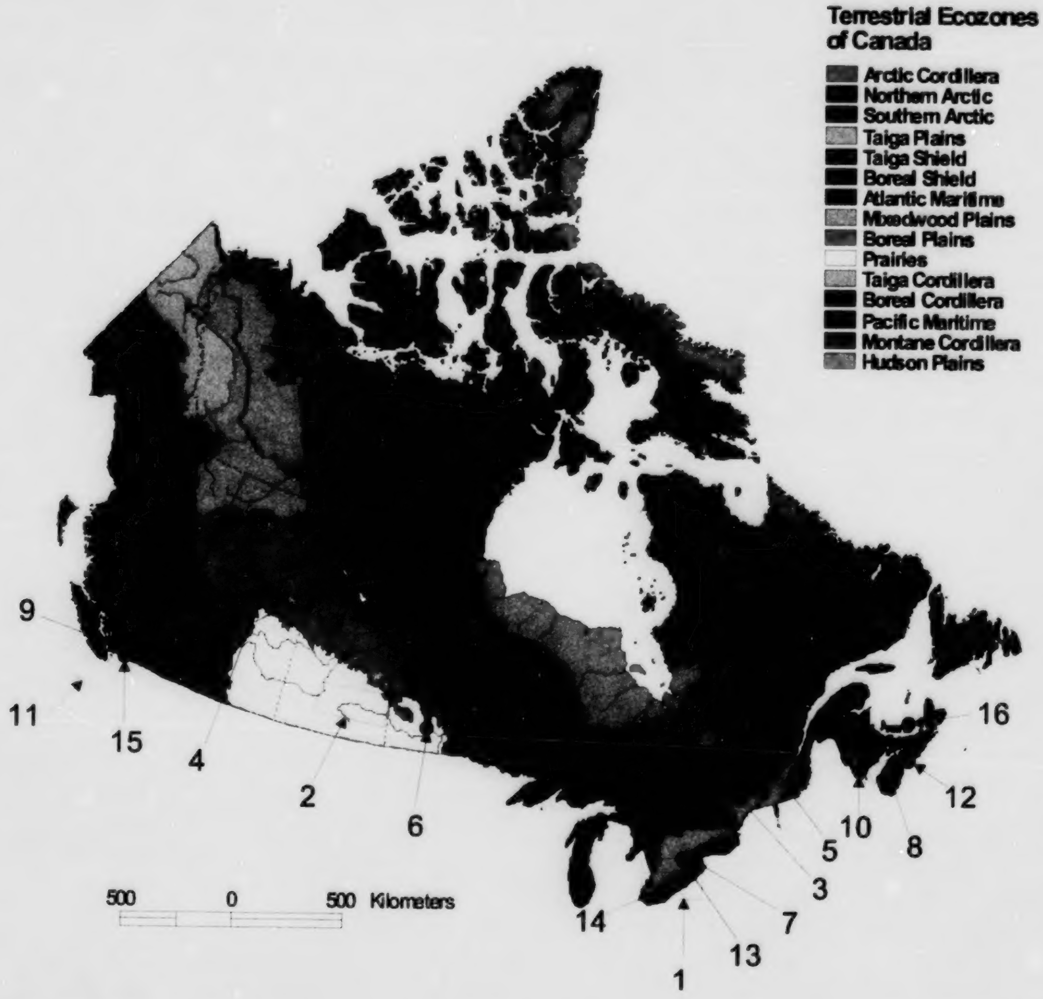
The 1995 Memorandum of Understanding (MOU) among the four federal natural resource departments (Environment, Natural Resources, Agriculture and Agri-Food, and Fisheries and Oceans) was identified as a good framework under which a scientific assessment of nutrients could be undertaken. As a result, a working group entitled *Nutrients in the Canadian Environment* was established under the MOU in 1997-98 to conduct the nutrient science assessment. The names of the members of the working group and their departmental affiliation are listed on page iii of this report. The MOU was renewed and expanded to include Health Canada effective January 1998.

1.4. Objectives of this Report

This report presents a synthesis of current knowledge of the effects of anthropogenic nutrients on the Canadian environment and the quality of life of Canadians. Nutrients for the purposes of this report follow the definition given in the 1999 CEPA (see text box, "Nutrients Defined"). The report evaluates:

- nutrient cycles in the environment;
- anthropogenic contributions to nutrient loading;
- enrichment responses to nutrient addition;
- toxic effects of nutrients;
- rationales for current practices for managing anthropogenic nutrients, and
- emerging issues related to the management of nutrients.

Case studies from a variety of ecozones are used to illustrate the effects of added nutrients on Canadian environments (Figure 1.1). The information is then used to identify the presence, extent and source of environmental problems caused by excess nutrients in Canada, and assess whether nutrients as a class, certain nutrients, or nutrient-containing products are problematic.


Terrestrial Ecozones of Canada
Arctic Cordillera
Northern Arctic
Southern Arctic
Taiga Plains
Taiga Shield
Boreal Shield
Atlantic Maritime
Mixedwood Plains
Boreal Plains
Prairies
Taiga Cordillera
Boreal Cordillera
Pacific Maritime
Montane Cordillera
Hudson Plains
9
11
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4
2
6
16
12
10
8
5
3
7
13
14
1
500 0 500 Kilometers

Figure 1.1. Locations of case studies presented in this report, 1998.

1	Lake Erie, ON	-effects of nutrient addition on a Great Lake (p. 64)
2	Qu'Appelle Lakes, SK	-effects of nutrient addition on a chain of prairie lakes (p. 67)
3	Lac Heney, QC	-effects of nutrient addition caused by aquaculture (p. 69)
4	Northern Rivers, AB	-effects of nutrient addition on a large northern rivers (p. 73)
5	Yamaska River, QC	-effects of nutrient addition from agriculture on a river (p. 75)
6	Delta Marsh, MB	-nutrient status of a prairie wetland (p. 79)
7	Cootes Paradise, ON	-effects of urban activity on a Great Lakes marsh (p. 80)
8	Bay of Fundy, NS	-nutrient cycling in a coastal salt marsh (p. 82)
9	Fraser River Estuary, BC	-effects of urban development and nutrients on a coastal marsh (p. 83)
10	Bay of Fundy, NS	-effects of marine aquaculture on coastal ecosystems (p. 85)
11	Georgia Basin, BC	-nutrient loading to the Strait of Georgia (p. 86)
12	Halifax Harbour, NS	-nutrient loading to Halifax Harbour (p. 88)
13	Cambridge, ON	-effects of nutrient loading from a domestic septic system (p. 92)
14	Southwestern ON	-effects of nutrients on amphibian (p. 118) and fish (p. 120) populations
15	Abbotsford Aquifer, BC	-effects of agricultural nutrients on ground water (p. 125)
16	Cardigan River, PEI	-links between nutrients and contaminated mussels (p. 128)

+1

2.0 Nutrient Cycles

Highlights

- Prior to industrialization and intensive agriculture, microbial fixation was the primary mechanism for converting nitrogen gas to forms of nitrogen that could be taken up by plants. Bioavailable phosphorus was added to soil and water mainly through the weathering of phosphorus-containing rock.
- Human activity has increased the production of bioavailable nitrogen and phosphorus.
- Municipal, industrial, and agricultural effluents have drastically increased the supply of phosphorus and nitrogen to freshwater and coastal ecosystems.
- Fertilizer production has increased the rate at which atmospheric nitrogen is fixed as ammonia and necessitated the mining of phosphorus-containing rock.
- Fossil fuel combustion causes nitrogen in the air to combine with oxygen in the air and produce nitric oxide (NO) and nitrogen dioxide (NO_2) that contribute to photochemical smog and acid rain production, and nitrous oxide (N_2O) that contributes to greenhouse warming.

Nitrogen and phosphorus, along with all other chemical elements, circulate in the biosphere in characteristic paths from the abiotic environment to organisms and back to the abiotic environment. These more-or-less circular paths are known as biogeochemical cycles and they describe the chemical forms, pathways and reservoirs that exist for a particular element. Biogeochemical cycles fall into two broad categories: a gaseous-type cycle where the atmosphere is a large reservoir, and a sedimentary-type cycle where the largest reservoir is mineral-bearing rocks. Nitrogen is an example of a complex gaseous-type cycle whereas P represents a simple sedimentary-type cycle. To understand the impacts of human activity on the transport, fate and effects of N, P, or any other element, it is first necessary to examine the biogeochemical cycle prior to extensive human alteration and then evaluate the changes in pathways and reservoirs resulting from human activity.

2.1. Nitrogen Cycle

The nitrogen cycle is essentially a process by which unreactive nitrogen gas (N_2) is converted to reactive forms that can be used by biological systems until conversion back to N_2 (Figure 2.2). The N cycle is unique in that, although 80% of the Earth's atmosphere is composed of N, most of this N is in the form of nitrogen gas (N_2) that cannot be used by the vast majority of organisms.

In the atmosphere, N exists primarily as nitrogen gas, nitrogen oxides (N_2O, NO and NO_2), and ammonia (NH_3). Nitrogen gas can be produced by the dissociation of nitrous oxide (N_2O) and destroyed as a result of its conversion to nitric oxide (NO) by lightning discharges (Figure 2.2). Nitrous oxide (N_2O) can also interact with a highly reactive oxygen atom to form NO and nitrogen dioxide (NO_2). These, in turn, react with water vapour to form nitric acid (HNO_3). Nitrogen oxides (NO and NO_2) can also be formed in a reaction involving ammonia and hydroxyl (OH^-) radicals. The two common nitrogen oxides, NO and NO_2, also cycle back and forth in a complex reaction involving the conversion of an oxygen molecule and a highly reactive oxygen atom to ozone (Figure 2.2).

Forms of Nitrogen in the Environment

Compound Name	Symbol	Environmental Location	State
Un-ionized Ammonia	NH_3	water, soil, air	aqueous
Ionized Ammonia, Ammonium ion	NH_4^+	water, soil, air	aqueous, gas
Nitrate	NO_3^-	water, soil, air	aqueous
Nitric acid	HNO_3	air	aqueous
Nitric oxide	NO	air	gas
Nitrite	NO_2^-	water, soil, air	aqueous
Nitrogen dioxide	NO_2	air	gas
Nitrogen gas	N_2	water, soil, air	gas
Nitrogen oxides (includes NO, NO_2, N_2O)		air	gas
$NO+NO_2$	NO_x	air	gas
Nitrous oxide	N_2O	water, soil, air	gas

The soil-plant pathways of the nitrogen cycle are dependent upon the conversion of unreactive nitrogen gas in the atmosphere to reactive N. Prior to industrial development, only two processes converted unreactive nitrogen gas to reactive N: biological nitrogen fixation (i.e., microbially-mediated production of ammonia) and lightning (which results in the production of nitric oxide, NO). Biological nitrogen fixation is a process whereby N_2 in the atmosphere is reduced to ammonia and incorporated directly into living tissue. A limited number of bacteria species possess the enzyme system necessary for this task. These include some of the blue-green algae (Cyanophyta) occurring in fresh waters, some soil bacteria, and the bacterial symbionts associated with the root nodules of legumes and alders. In addition to biological fixation, NO produced by lightning discharges can be converted to nitrate. Nitrate along with ammonia is deposited on the Earth's surface in snow or rain (wet deposition) or as particles such as nitrate or ammonium salts (dry deposition).

Once N is fixed, it can be converted to organic nitrogen within the fixing organism or lost from the organism as ammonium or urea. Under aerobic conditions, specialized bacteria species convert ammonium to NO_2^- and NO_3^-, a process known as nitrification. All marine, freshwater and terrestrial plants assimilate nitrogen as NH_4^+ or NO_3^-, with the exception of certain bog species that can only use NH_4^+. This assimilation process involves N uptake through the roots or other cells, assimilatory reduction of NO_3^- to NH_3, and metabolic conversion into tissue. Grazing animals ingest the organic nitrogen in plants and, in turn, are consumed by carnivores. Animals excrete nitrogenous wastes in the form of urea, uric acid or faeces, which are converted microbially back to NH_3. Organic nitrogen in dead organic material undergoes microbially-mediated decomposition (humification) in which carbon and nitrogen compounds are broken down, ultimately releasing forms of N than can be taken up by plants.

Prior to the development of agriculture and permanent settlement, the N in soils, water and plant and animal material was returned to the atmosphere largely through denitrification, the bacterially-mediated conversion of NO_3^- to N_2O and N_2 by microbes in soils and water under conditions of low oxygen (Galloway et al. 1995). Ammonia was also released by volatilization (a process by which molecules escape from a liquid phase to a gaseous phase) from vegetation, soils and animal waste, as a by-product of decomposition, and as a result of biomass burning. A comparatively smaller quantity of N

+5

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Nitric acid	HNO_3	air	aqueous
Nitric oxide	NO	air	gas
Nitrite	NO_2^-	water, soil, air	aqueous
Nitrogen dioxide	NO_2	air	gas
Nitrogen gas	N_2	water, soil, air	gas
Nitrogen oxides (includes NO, NO_2, N_2O)		air	gas
$NO+NO_2$	NO_x	air	gas
Nitrous oxide	N_2O	water, soil, air	gas

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The Role of Nitrogen in Air Pollution

Urban Smog

Ground-level, or tropospheric, ozone is a major constituent of "urban smog" and is of particular concern in several of Canada's major urban areas during spring and summer. Although chemical reactions producing smog are complex, the main feature is the dissociation of nitrogen dioxide (NO_2), introduced into the air largely from automotive and industrial emissions, to yield an oxygen atom ($O^{\cdot}$) and NO in the presence of sunlight. The resulting oxygen atom combines with an oxygen molecule (O_2) to form ozone (Figure 2.1). Ozone production cannot exceed NO availability unless an alternative means of regenerating NO_2 exists. This alternative is provided by free radicals in hydrocarbon-contaminated air. These free radicals combine with NO to regenerate NO_2, which then dissociates to form more ozone. During the day, ozone concentrations increase until the rate of ozone destruction equals the photochemical production of ozone. During the night, urban smog dissipates because dissociation of NO_2 only occurs in sunlight.

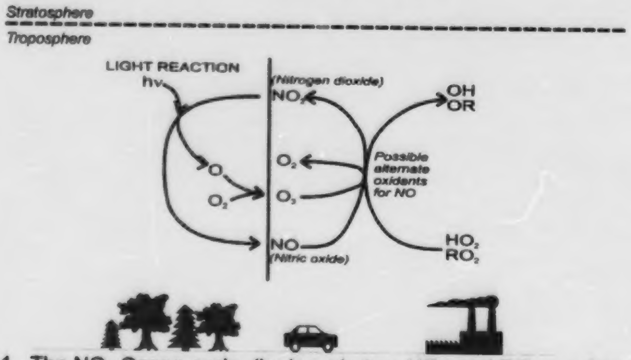

Figure 2.1. The NO_x-Ozone cycle. (hv is a photon; $HO_2^{\cdot}$ and $RO_2^{\cdot}$ are free radicals)

Ground-level ozone causes inflammation of the respiratory system that can persist for up to 18 hours after exposure ceases. It can also aggravate existing heart and lung conditions and may heighten the sensitivity of asthmatics to allergens.

Greenhouse Warming

Water vapour, carbon dioxide, methane and nitrous oxide (N_2O) regulate the Earth's temperature by reducing the rate at which radiation escapes into space. Without this greenhouse effect, the Earth would be about 33°C colder and inhospitable to life. Greenhouse gas concentrations have been increasing steadily since the industrial revolution, almost certainly because of human activity. In Canada, scientists predict an increase in annual mean temperatures in some regions of between 5 and 10°C over the next 100 years. Increased incidences of extreme weather events are also predicted.

Nitrous oxide (N_2O) is a potent greenhouse gas with a warming potential over 100 years about 310 times that of CO_2. It is added to the atmosphere from fertilizers, biomass burning, fossil fuel combustion and industrial activities. In the stratosphere, it can be destroyed by photolysis or by reaction with oxygen radicals where it can catalyze the destruction of stratospheric ozone. Ozone in the upper atmosphere filters out UV radiation from the sun, so its depletion results in increased intensity of UV radiation reaching the Earth's surface.

Acid Rain

Acid rain is caused by the conversion of pollutants such as sulfur dioxide (SO_2) and NO_X (i.e., nitric oxide (NO) and nitrogen dioxide (NO_2)) into sulfuric acid and nitric acid in the atmosphere. Dilute forms of these acids fall to earth as precipitation or are deposited as acid gas or dust. A major source of both SO_2 and NO_X is combustion of fossil fuels in motor vehicles, residential and commercial furnaces, and industrial boilers and engines. Once released into the atmosphere, these pollutants can be transported long distances. For example, more than 50% of acid rain in eastern Canada comes from sources in the United States.

Acid precipitation to lakes causes loss of major elements of the food web, including crustaceans, plankton, insects and, ultimately, fish. Another effect of acidification is to increase the toxicity of heavy metals (e.g., aluminium). In forest ecosystems, acid precipitation can damage the surfaces of tree leaves and needles, reduce a tree's ability to withstand cold, and inhibit plant germination and reproduction. However, the nitrogen in acid precipitation can also stimulate forest growth (see Chapter 4.1). Ultimately, changes in soil chemistry caused by acid rain may cause a decline in soil nutrients, resulting in slower growth rates and increased death rates in trees.

was released as NO, a secondary product of nitrification (bacterially-mediated conversion of NH_4^+ to NO_3^-).

In the present world of industrialization and intensive agriculture, both fixation and emission rates of N have changed compared to pre-industrial times. Currently, three anthropogenic activities produce reactive nitrogen (in addition to two naturally-occurring activities): combustion of coal, petroleum products and natural gas, which adds NO to the atmosphere; fertilizer production, which fixes nitrogen gas as ammonia; and cultivation of legumes and other crops, which enhances biological nitrogen fixation beyond background levels. These activities have more than doubled the global rate of N fixation compared to pre-industrial times (Galloway et al. 1995; Vitousek et al. 1997).

The importance of the components of the soil-plant pathway of the nitrogen cycle has also changed from pre-agricultural to present times. In the pre-agricultural world, the breakdown of soil organic matter by soil micro-organisms supplied most of the N for plant growth. Currently, chemical fertilizers and manure are a significant source of N to agricultural crops and, in some cases, forests managed for timber harvest. In addition to enhancing fixation, human activity has liberated long-term biological storage pools through the drainage of wetlands and consequent oxidation of their organic soils. Human activity has also redistributed N between terrestrial and aquatic environments and among ecosystems as a result of municipal and industrial wastewater discharges, and runoff or leaching from land managed for urban development, forestry or agriculture. Because growth of most terrestrial vegetation increases in response to added N, the additional N supply along with its redistribution has likely increased plant growth on a global basis and thus the quantity of organic carbon stored in terrestrial ecosystems worldwide (Vitousek et al. 1997). However, for specific forested ecosystems, controversy remains as to whether productivity gains due to increased N deposition, along with a warming climate and elevated CO_2 levels in the atmosphere, have offset carbon losses caused by timber harvest, fire, and insect-induced mortality. The additional N supplied to terrestrial systems has also moved into ground and surface waters, thereby contributing to overproduction of aquatic plants (particularly in certain north-temperate coastal waters) and to exceedances of nitrate and ammonia guidelines for the protection of aquatic life (particularly near sites where human waste or manure is poorly managed).

As a result of increased fixation, liberation and redistribution of N due to human activity, the composition of atmospheric nitrogen gases has changed and this alteration has contributed to acid rain, urban smog and global warming. Ammonia emissions from fertilized fields and livestock waste, and biomass burning have become major atmospheric sources of N (Galloway et al. 1995). Likewise, fertilized soils and biomass burning have also increased NO and N_2O emissions to the atmosphere. These emissions, as well as NO emissions from fossil fuel combustion, have elevated atmospheric concentrations of nitrogen oxides. The end production of NO oxidation is nitric acid, a principal component of acid deposition (see “the Role of Nitrogen in Air Pollution” text box). As well, increased levels of nitric oxide and nitrogen dioxide react with oxygen and free radicals of hydrocarbons to produce ground-level ozone, a phenomenon responsible for urban smog. Finally, increased nitrous oxide levels contribute to global warming by absorbing infrared radiation.

2.2. Phosphorus Cycle

In contrast to N, phosphorus does not exist naturally in the elementary state (i.e., as P_4). Also unlike N, where the largest reservoir is the atmosphere, the largest P reservoir is rock (e.g., apatite). All P in the

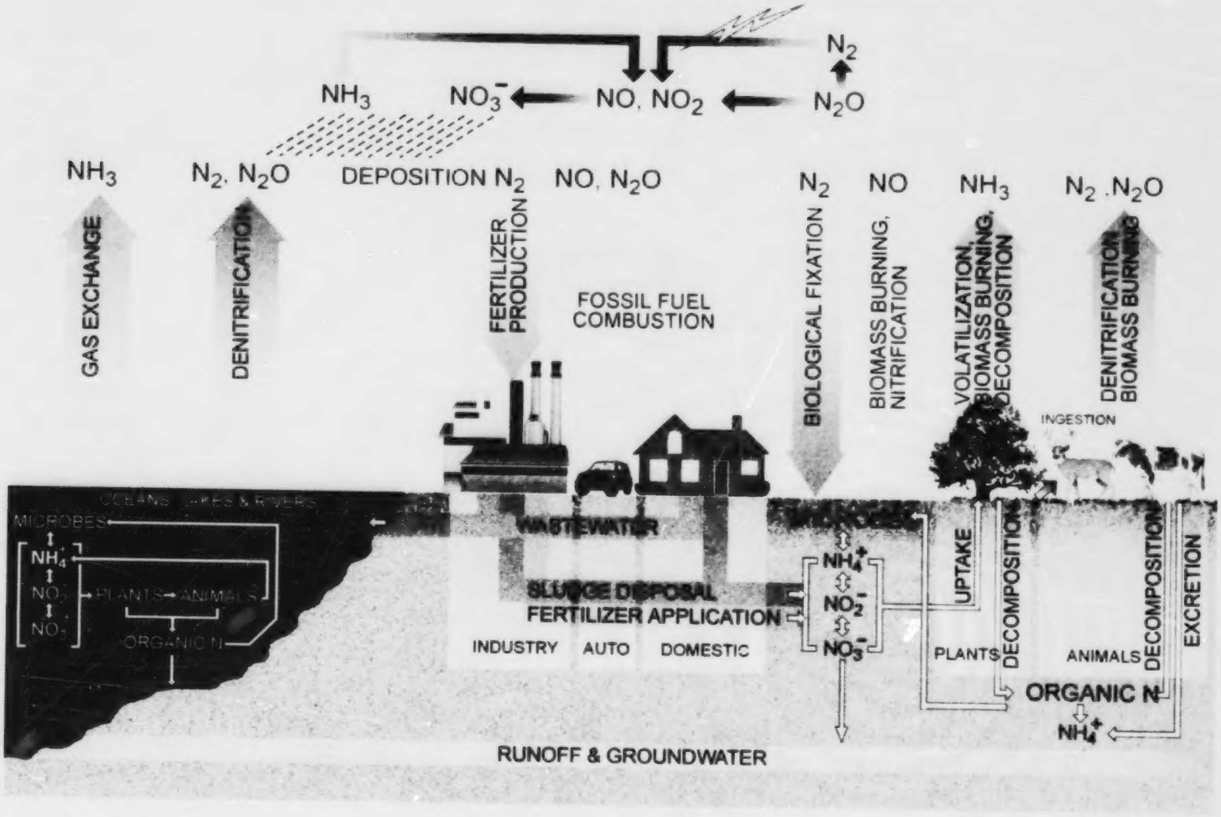
N2
NH3
NO3-
NO, NO2
N2O
NH3
N2, N2O
DEPOSITION
N2
NO, N2O
N2
NO
NH3
N2, N2O
GAS EXCHANGE
DENITRIFICATION
FERTILIZER PRODUCTION
FOSSIL FUEL COMBUSTION
BIOLOGICAL FIXATION
BIOMASS BURNING, NITRIFICATION
VOLATILIZATION, BIOMASS BURNING, DECOMPOSITION
DENITRIFICATION BIOMASS BURNING
INGESTION
OCEANS, LAKES & RIVERS
MICROBES
NH4+
NO2-
NO3-
PLANTS → ANIMALS
ORGANIC N
WASTEWATER
SLUDGE DISPOSAL
FERTILIZER APPLICATION
INDUSTRY
AUTO
DOMESTIC
MICROBES
NH4+
NO2-
NO3-
UPTAKE
DECOMPOSITION
PLANTS
ANIMALS
DECOMPOSITION
EXCRETION
ORGANIC N
NH4+
RUNOFF & GROUNDWATER

Figure 2.2. The nitrogen cycle.

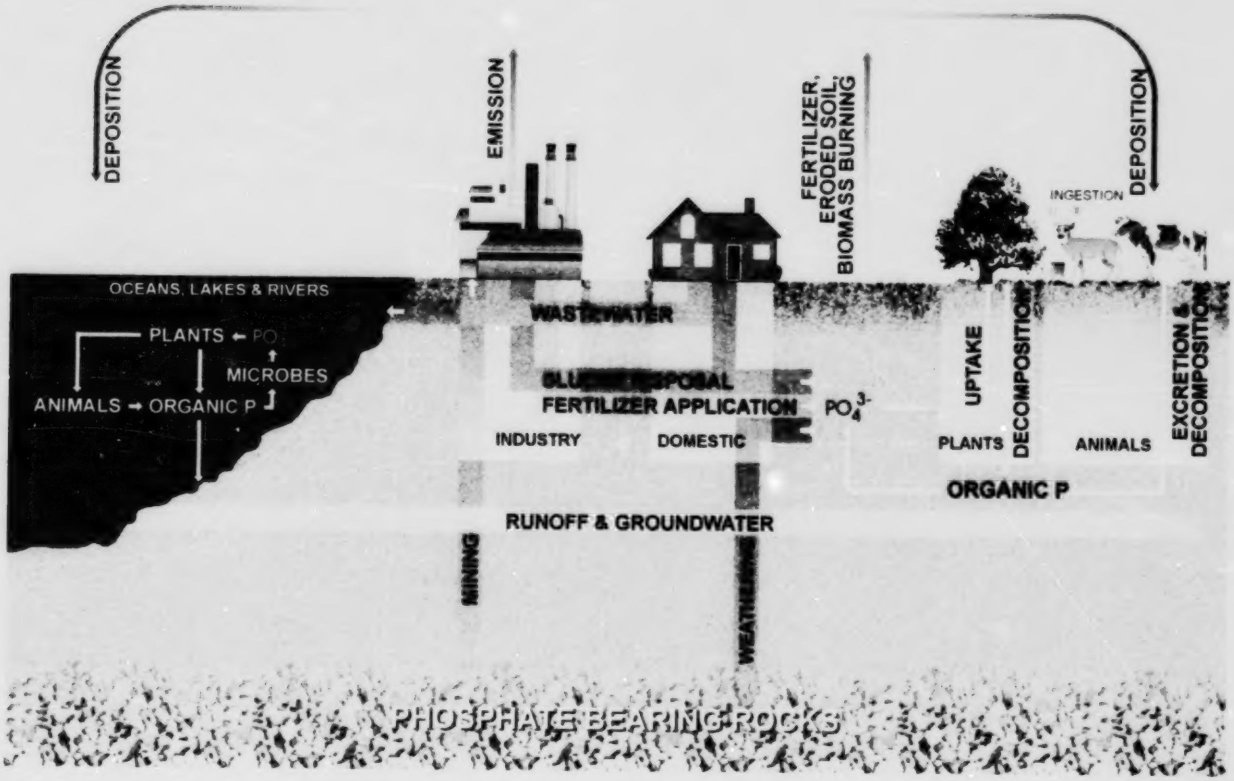
DEPOSITION
EMISSION
FERTILIZER, ERODED SOIL, BIOMASS BURNING
DEPOSITION
INGESTION
OCEANS, LAKES & RIVERS
PLANTS ← PO
MICROBES
ANIMALS → ORGANIC P
WASTEWATER
SLUDGE DISPOSAL
FERTILIZER APPLICATION
PO_4^{3-}
INDUSTRY
DOMESTIC
UPTAKE
DECOMPOSITION
PLANTS
ANIMALS
EXCRETION & DECOMPOSITION
ORGANIC P
RUNOFF & GROUNDWATER
MINING
WEATHERING
PHOSPHATE BEARING ROCKS

Figure 2.3. The phosphorus cycle.

Forms of Phosphorus

Phosphorus occurs in the environment as inorganic phosphate and incorporated into organic compounds. The separation of phosphorus into its various forms is defined analytically; these analytical groupings have been selected so they may be used for interpretative purposes.

Total phosphorus (TP) is a measure of all forms of P in a water, soil, or air sample (Figure 2.4). Analysis for TP involves a strong digestion (in order to release P from organic matter) followed by a colourimetric determination of orthophosphate concentration ("colourimetry").

Water samples are typically analyzed for TP as well as two forms of dissolved P: total dissolved P (TDP) and soluble reactive P (SRP). When a water sample is filtered through a 0.45-µm membrane, the dissolved P forms are retained in the filtrate (Figure 2.4). TDP analysis involves a strong digestion of the filtrate followed by colourimetry whereas SRP analysis entails only colourimetry.

In soils, P is bound to soil particle surfaces. Its availability to plants for growth is assessed using a variety of extractions of differing intensity.

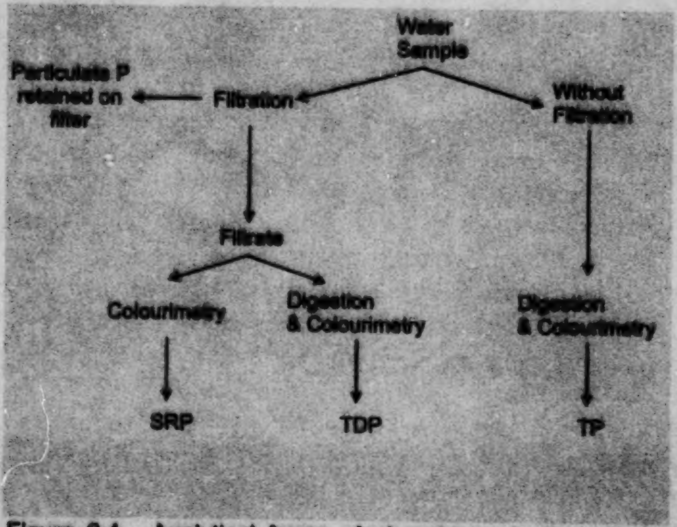

Figure 2.4. Analytical forms of phosphorus in the aquatic environment.

atmosphere is derived from sources on Earth (e.g., from erosion, fertilizer drift, industrial emission, etc.). Finally, unlike N, P exists in only one environmentally reactive form, orthophosphate (PO_4^{3-}).

Prior to industrialization and intensive agriculture, P was added to soil as the orthophosphate ion (PO_4^{3-}) only through the weathering of parent material (Figure 2.3). Phosphorus is taken up from the soil mainly as orthophosphate through plant root systems or other cells. Plants are grazed and animals excrete excess P as phosphorus salts in urine and faeces. This P is then either taken up directly or converted from organic phosphorus to phosphate ion by a wide range of bacteria and fungi (Ricklefs 1990). The decomposition of organic matter also returns inorganic phosphorus to the soil. Phosphate is extremely reactive and binds with many cations (i.e., iron, aluminum, and calcium) to form relatively insoluble compounds. This binding results in the storage of phosphate in poorly labile forms in both terrestrial and aquatic ecosystems.

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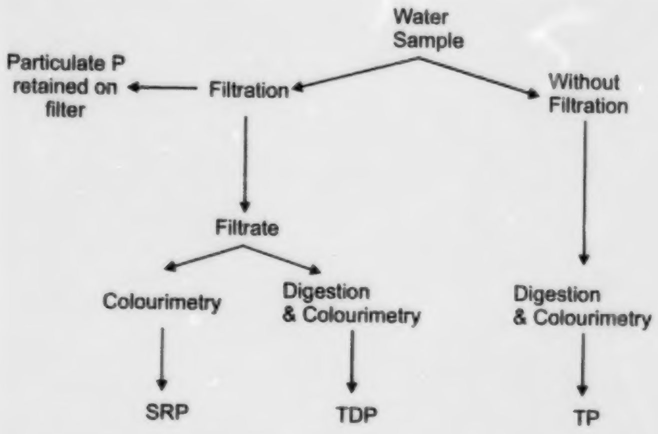

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In the modern world of industrialization and intensive agriculture, mining of phosphorus-containing rock to produce fertilizer has increased the supply of P to soils. These fertilizers are applied in locations where the rock is naturally lower in P and at rates that far exceed weathering. In addition, municipal and industrial sources contribute P to water (wastewater discharges), air (stack emissions) and soil (landfills, sludge application). Runoff from land managed for urban development, forestry or agriculture is also a major P source for aquatic vegetation. The result is that P is distributed around the globe as P-bearing rock, fertilizer, livestock feed and agricultural produce, and as waste emissions to water and the atmosphere. Because plant growth in most freshwater ecosystems is constrained by a shortage of P, the addition of P, albeit dilute, can cause significant increases in the productivity of lakes and rivers.

2.3. Conclusions

Human activity has increased the supply of bioavailable forms of both N and P. In the case of N, atmospheric nitrogen is converted to fertilizer and applied to agricultural fields to grow crops, which are then transported to urban areas. Urban sewage, which is high in N, is discharged to surface waters because it may be contaminated or too costly (because of long transport distances) to apply to agricultural land. Fossil fuel combustion (by industries, in automobiles, and for household heating) enhances nitrogen oxide production as the high temperatures associated with fossil fuel combustion cause N in the air to combine with oxygen in the air. As a result, the amount of available N has more than doubled since the 1940s, with human activities contributing 210 million tonnes per year to the global supply of N compared to only 140 million tonnes generated per year during pre-industrial times (Vitousek et al. 1997). Human alterations to N cycling have increased concentrations of the potent greenhouse gas N_2O globally; increased concentrations of nitrogen oxides driving the formation of photochemical smog in large regions of the Earth; increased the supply of N to vegetation; increased transfer of N to freshwater and coastal ecosystems; and enhanced surface water acidity through the addition of nitrate (Vitousek et al. 1997).

Similar changes have occurred in the phosphorus cycle. Weathering of phosphate-bearing rocks as a source of P is now overshadowed by mining activities, with approximately 17 million tonnes of P now mined each year. Manufactured fertilizer is transported to agricultural fields far from the rock source. Agricultural crops, in turn, are transported as feed to areas with high livestock concentrations and to urban areas. Urban sewage is discharged to surface waters because it may be too contaminated or too costly to apply to agricultural land. The environmental consequences of human alterations to P and N cycling are serious and long term.

3.0 Anthropogenic Sources of Nitrogen and Phosphorus

Highlights

- In 1996, 73% of Canadians were served by municipal sewer systems. An additional 25% relied on septic beds for sewage treatment. The remaining 2% were likely serviced by lagoons.
- Human waste is the largest source of nitrogen and phosphorus to municipal wastewater treatment plants.
- In 1996, at least 94% of the sewage collected by sewers received primary treatment or better. The majority of the sewage collected in the interior of the country, from Alberta to Ontario, receives secondary treatment or better. For Canadians living on the coasts, discharge of untreated sewage is common.
- Phosphorus loading from municipal wastewater treatment plants in Canada has decreased by 37% from 1983 to 1996. N loads, however, have increased by 17%.
- Agricultural crop production in Canada has doubled in the last 50 years. This increase is due to improved crop varieties, improved crop management, pesticide use, and increased use of manure and fertilizers.
- Fertilizer, manure, and legume nitrogen fixation are the major sources of nutrients to agricultural land. In 1996, fertilizer was typically applied to cropland in Canada at rates of 60 to 86 kg/ha nitrogen and 10 to 33 kg/ha phosphorus. Manure was typically applied at rates of 114 to 301 kg/ha N or 38 to 184 kg/ha P.
- Agricultural activities contribute about 91% of the ammonia from Canadian sources to the atmosphere. Agricultural runoff and leaching is also a source of nutrients to surface and ground waters.
- The majority of the 204 t P/yr and 956 t N/yr added to Canadian inland waters by aquaculture operations is the result of uneaten food fragments and metabolic waste.
- Atmospheric deposition supplies, on average, 3.4 kg/ha/yr dissolved inorganic N east of the Manitoba-Ontario border and 0.8 kg/ha/yr west of this border. Wet and dry deposition of P ranges from 0.01 to 0.7 kg P/ha/yr for all of Canada.

Nitrogen and phosphorus enter the environment as a result of both natural processes and human activity (see Chapter 2). The largest reservoir of N exploited by humans is nitrogen gas (N_2) in the atmosphere. Most manufactured N compounds are made from atmospheric nitrogen gas. The major manufactured N-containing product is fertilizer. Ammonia is also used as a nutrient in fermentation processes in food and beverage industries, and for production of some synthetic fibres (e.g., nylon) (Kettrup and Hüppe 1988). Derivatives of hydrazine (N_2H_4) are used as chemotherapeutic agents, particularly in the treatment of leprosy and tuberculosis. Nitrate and nitrite compounds are also used in the production of commercial explosives, azo dyes and pharmaceuticals.

Industrial P chemicals are made from phosphate rock transformed into phosphoric acid by either smelting or treatment with acid. Of the phosphate produced by the world's industry today, about 80 to 85% is used in fertilizers. The next largest user is the detergent industry. Until the 1950s, most detergents were soap-based products made from animal fat and lye (sodium hydroxide). In 1947, the

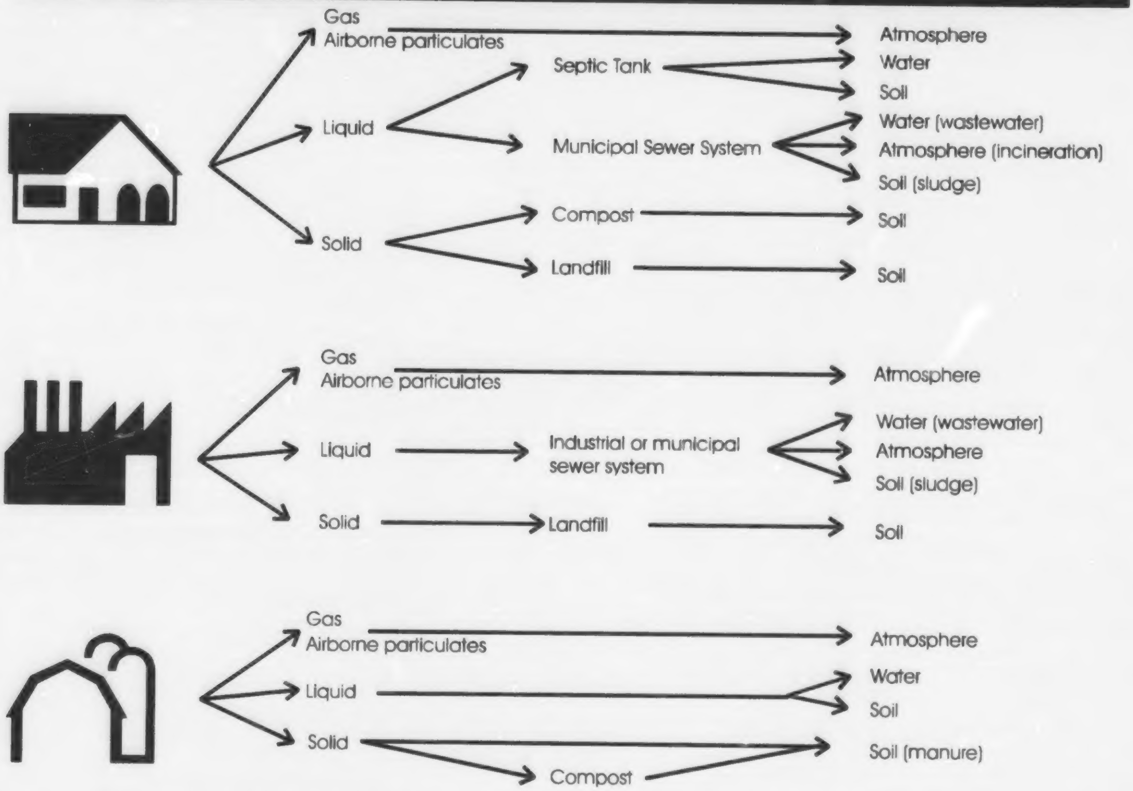
Gas
Airborne particulates
Atmosphere
Septic Tank
Water
Soil
Liquid
Municipal Sewer System
Water (wastewater)
Atmosphere (incineration)
Soil (sludge)
Compost
Soil
Solid
Landfill
Soil
Gas
Airborne particulates
Atmosphere
Liquid
Industrial or municipal sewer system
Water (wastewater)
Atmosphere
Soil (sludge)
Solid
Landfill
Soil
Gas
Airborne particulates
Atmosphere
Liquid
Water
Soil
Solid
Soil (manure)
Compost

Figure 3.1. Household, industrial and agricultural sources and fate of N and P in the environment.

first synthetic detergents were introduced and gained wide acceptance because of their improved cleaning performance. The basis of these new formulations was sodium tripolyphosphate, used as a "builder" to soften water and optimize washing conditions for other active ingredients. Until the late 1980s, sodium tripolyphosphate was used almost exclusively as the builder in laundry detergents. Because of effects on water quality, other builders have been introduced in recent years although shortcomings in the performance of these builders, as compared to sodium tripolyphosphate, require additions of other chemicals (CEEP 1998).

Phosphate is also used in the manufacture of animal feed supplements because of its nutritive value. Food-grade phosphates are used in food products such as dairy, meat and bakery products, and in soft drinks. For example, phosphate compounds are used as leavening agents by bakers and as a polishing agent in toothpastes. Other industrial applications include the manufacture of flame retardants, in the treatment of metal surfaces to prevent rusting ("phosphatizing"), and as a bath for the electropolishing of stainless steel articles (Kettrup and Hüppe 1988).

In short, nitrogen and phosphorus are integral components of our daily lives. Nitrogen and phosphorus fertilizers make it possible to meet much of the world's food demands. Many of the products that we use contain N or P or have required N or P during their manufacture. In addition, N and P are vital components of growth and metabolism for all animals, including humans. In humans, P accounts for 1.1 to 1.2 g/kg body weight, most (85%) of which occurs in bone and teeth. Nitrogen forms the foundation of amino acids which act as the building blocks of proteins and DNA and thus plays an important role in the regulation of essential physiological reactions (West et al. 1966). A consequence of the transformation of N and P from natural reservoirs to products used in homes and industries and to food consumed by humans and other animals is the undesirable loss of nutrients to the environment. Transformation of N and P by biological and industrial processes are not completely efficient such that nutrients are lost at every stage in the process.

The purpose of this chapter is to identify and quantify the human derived, or anthropogenic, sources of P and N to the Canadian environment. The major anthropogenic sources are municipal, industrial and agricultural losses. For these sectors, nutrients may be released to the atmosphere as gases or airborne particulates; to surface or ground waters in the form of wastewater discharges and runoff, in the case of land-based operations such as agriculture or forestry operations; and to the soil as solid waste (Figure 3.1). In this chapter, the release of nutrients to surface and ground waters, to the atmosphere and to soils is considered for municipal, industrial, agricultural, aquacultural and forestry sectors.

3.1 Municipal Waste

Municipal waste consists of liquid waste, also known as wastewater or sewage, and solid waste or the garbage collected from households and businesses.

Municipal wastewater is a complex mixture of suspended solids, micro-organisms, debris and a variety of chemicals derived from both household and industrial sources (New Brunswick Environment 1982; Birtwell et al. 1983; OME 1988). Almost all households, office buildings and small to medium-size industries discharge their wastewater to a municipal sewer system. Large industries (pulp mills, mining operations, large manufacturing plants, etc.) often independently treat and discharge their wastewater; discharges from industries with provincial operating permits are presented in Section 3.2.

Municipal wastewater is conveyed from households, businesses and roadways by a complex series of sewers. There are two types of sewer systems:

- Separate systems, comprising sanitary sewers that carry raw sewage from homes and businesses to wastewater-treatment facilities, and storm sewers that carry storm runoff from streets, parking lots, and roofs through pipes and ditches and eventually into streams, lakes or coastal waters,
- Combined sewer systems that carry raw sewage, with or without storm water, to wastewater treatment facilities during periods of no or low precipitation, but discharge excess raw sewage and storm water into receiving waters during periods of high rainfall or snowmelt when their flow capacity is exceeded.

Cities have either a separate sewer system (consisting of independent storm sewers and sanitary sewers) or a combined sewer system. In rural settings, low population densities do not make sewer systems necessary or economically feasible. Instead, rural households typically discharge sewage to septic disposal systems or holding tanks.

Effluents from Municipal Wastewater Treatment Plants

In 1992, there were approximately 2 800 municipal wastewater treatment plants (MWTPs) in Canada. The percentage of Canadians served by wastewater treatment has increased in recent years. Surveys conducted by Environment Canada (1996a) showed that 73% of Canadians were served by municipal sewer systems as of 1996. The remaining 27% (8 million Canadians) were in villages (with populations less than 1000) or rural settings and were largely served either by septic disposal systems (25%) or

lagoons (2%). Of those hooked up to municipal sewers, 94% (20.7 million Canadians) were served by wastewater treatment (primary or better) in 1996 compared with 85% (17.4 million Canadians) in 1991 (Figure 3.2). The remaining 6% (1.3 million Canadians) were serviced by sewage collection structures not connected to wastewater treatment facilities but that discharged untreated sewage directly into lakes, rivers or oceans (Environment Canada 1996a).

Defining Sewage Treatment Types

Treatment Type	Description
Primary	The mechanical screening and sedimentation of sewage, to decrease influent biochemical oxygen demand (BOD) by 20 to 30% and total suspended solids (TSS) by about 60%.
Secondary	The mechanical aeration of sewage, to encourage biological degradation of soluble organic matter, followed by sedimentation of solids to decrease influent BOD and TSS by 80 to 95%.
Tertiary	The additional treatment by sand filtration or a polishing lagoon after secondary treatment to achieve higher TSS and BOD removal.
Lagoons/ Waste Stabilization Ponds	The treatment of sewage by biological processes in one or a series of relatively shallow basin(s). This process is commonly used in small communities and produces effluent equivalent to secondary treatment.
Phosphorus removal	A process in which either iron or aluminum solution (i.e., alum) is added to the sewage to reduce effluent TP concentrations. TP removal can be added at any stage of sewage treatment.

Definitions after OMEE 1993.

The level of sewage treatment is improving in Canada as more municipalities upgrade their wastewater treatment facilities. In 1996, tertiary treatment (largely advanced P removal) was provided to 38% of the municipal population, up from 36% in 1991 (Figure 3.3). Secondary treatment (including both mechanical systems, such as activated sludge methods process or percolating filters, and non-mechanical systems such as lagoons) was provided to 34% of the population, up from 29%,

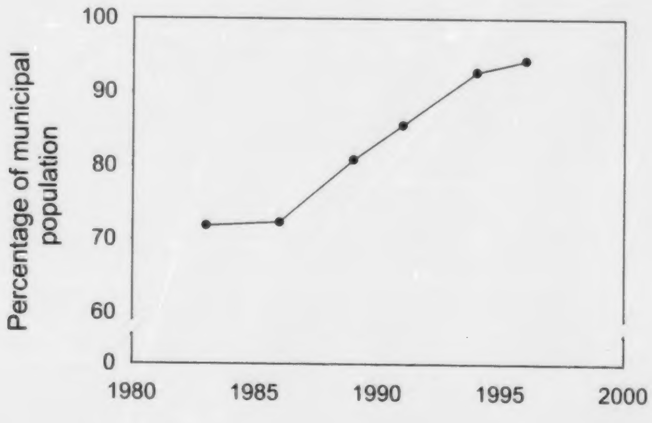

Figure 3.2. The proportion of Canada's population with municipal wastewater treatment, 1983-1996. Data are for communities with populations greater than 1 000 served by sewers (Environment Canada 1996a).

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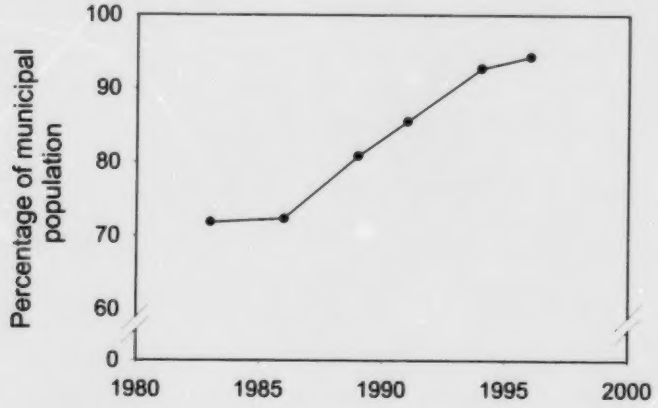

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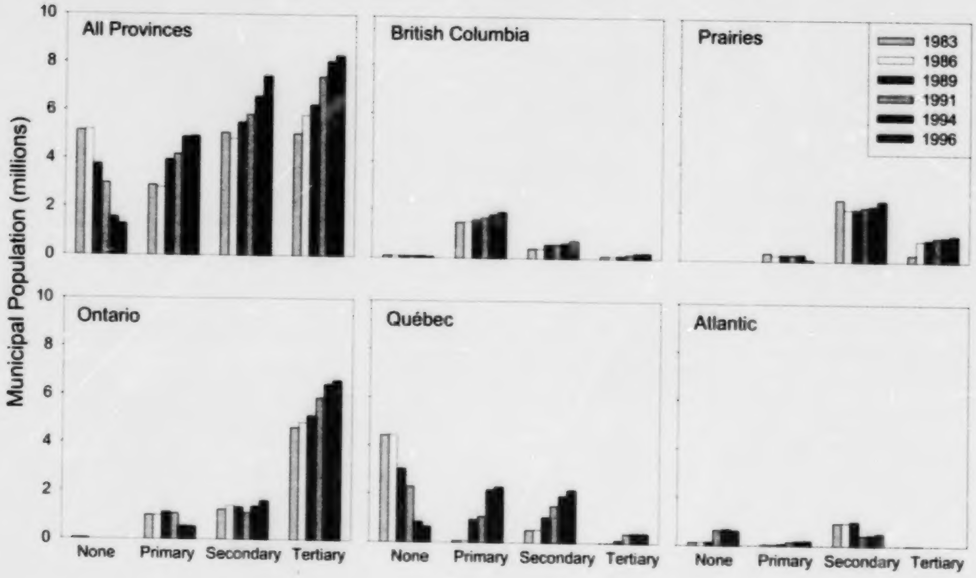
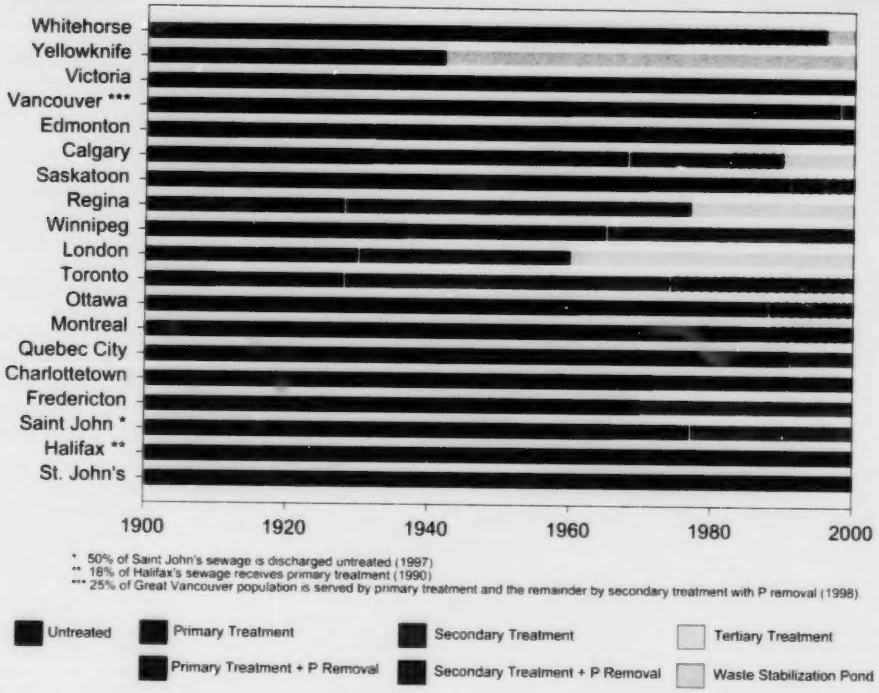


Figure 3.3. The number of Canadians in each region of Canada, excluding the Territories, served by sewage treatment, 1983-1996. Data are for communities with populations greater than 1 000 served by sewers (Environment Canada 1996a). Territorial data are not presented, as the small numbers are not distinguishable at this scale. Communities served by lagoons or by secondary treatment have been pooled as lagoons produce effluent equivalent to secondary treatment.

and primary treatment was provided to 22%, up from 20%. Much of this change occurred in Québec where the population served by wastewater treatment increased from 2 to 75% between 1980 and 1991 (MEFQ 1995).

The level of wastewater treatment varies greatly across Canada (Figure 3.3). Most of the population of British Columbia is served by primary treatment; however, this figure is an overestimate because Victoria's sewage treatment consists of screening through a 6-mm wire mesh but not sedimentation. Only slight increases in secondary and tertiary treatment in British Columbia have occurred in the past five years. In the Prairie Provinces, secondary treatment and tertiary treatment serve most of the population. Ontario's population is largely served by tertiary treatment, with substantial increases in this level of service since 1983 in response to programs to clean up the Great Lakes. In Québec, a mix of primary or secondary treatment serves most of the population, many of these upgrades having occurred in the past 10 years. In the Atlantic Provinces, more than half of the population is served by sewer systems releasing untreated wastewater directly into estuarine or coastal waters.

The variation in wastewater treatment across Canada is also evident in the current and historical status of service provided by major cities (Figure 3.4). Waste stabilization ponds (lagoons) currently serve the comparatively small populations of Whitehorse and Yellowknife. The two major West Coast cities, Vancouver and Victoria, have a long history of no sewage treatment. Victoria implemented screening in 1989 to remove large floatables such as logs and plastics from the sewage. Vancouver added primary treatment in 1961 followed, in 1998, by upgrades to secondary treatment for two of the city's three MWTPs. All the major Prairie cities (Edmonton, Calgary, Saskatoon, Regina, and Winnipeg)



* 50% of Saint John's sewage is discharged untreated (1997)
** 18% of Halifax's sewage receives primary treatment (1990)
*** 25% of Great Vancouver population is served by primary treatment and the remainder by secondary treatment with P removal (1998).

Figure 3.4. The history of sewage treatment in major Canadian cities, 1900-2000. The dates represent the beginning of upgrades to the treatment plants and not the dates of 100% implementation of the treatment. Data were obtained through consultation with each city's engineering or utilities department.

provide at least secondary treatment, with Regina, Saskatoon and Calgary also undertaking advanced P removal. In Ontario, Toronto and Ottawa have secondary treatment with advanced P removal; London has tertiary treatment. In Québec, Montreal's sewage receives primary treatment with advanced P removal whereas Québec City's sewage receives secondary treatment. In the Atlantic Provinces, many cities discharging sewage to the ocean (e.g., St. John's, Halifax, Charlottetown) provide no treatment or only primary treatment. Fredericton provides secondary treatment, whereas only half of the sewage from Saint John receives secondary treatment and the other half is discharged untreated.

Wastewater

In 1991, approximately 4 300 million cubic metres of municipal wastewater was discharged to Canadian lakes, rivers and coastal waters (Statistics Canada 1994). Although not all MWTPs measure the composition of their effluents, nutrient loads can be estimated from information on the average per capita nutrient load for the various levels of treatment. Data from a comprehensive study of municipal wastewater discharges by treatment type in Ontario in 1991 were used to calculate an average influent total P load of 3.38 g per capita per day as well as the average effluent total P load and removal efficiency for each level of sewage treatment (Table 3.1). To estimate the MWTP load for total P, the influent P load (3.38 g/capita/day) was then multiplied by the population served by each level of sewage treatment (Figure 3.3) and the removal efficiency for each sewage type (Table 3.1). Information on N loads was not collected as part of the Ontario survey. We therefore used an influent load of 10 g total N per capita per day based upon Tchobanoglous and Burton's (1991) estimate of 12 g N per capita per

Table 3.1. Total phosphorus load in the final effluent and removal efficiency for various levels of wastewater treatment. (Values were calculated from data presented in a 1991 survey of Ontario MWTPs (OMEE 1993). Removal efficiency was calculated as the difference between the influent and effluent load, expressed as percent of the influent load.)

Treatment Type	P Removal ?	Number of Facilities Sampled	Effluent Total Phosphorus Load (g/capita/d)	Total Phosphorus Removal Efficiency (%)
Primary	no	9	1.71	36.3
	yes	19	0.75	75.5
	Average	**28**	**1.06**	**62.9**
Secondary	no	46	1.03	59.0
	yes	137	0.42	88.4
	Average	**183**	**0.58**	**81.0**
Lagoons	no	45	0.78	65.5
	yes	76	0.20	92.5
	Average	**121**	**0.42**	**82.4**
Tertiary	no	2	1.02	58.7
	yes	33	0.15	94.7
	Average	**35**	**0.20**	**92.7**

day (which equates to delivery to the MWTP of 350 L/day at 35 mg/L TN) minus a 10% loss during sewage treatment (assuming no advanced treatment for N removal). The MWTP load for total N was estimated by multiplying the influent N load (10 g/capita/day) by population (Figure 3.3.).

In 1996, 5.6 thousand tonnes of P (as total P) was released to lakes, rivers and coastal waters in Canada from MWTPs (Figure 3.5). This discharge represents a 37% decrease since 1983 and a 20% decrease from 1991 loads. The total P load from Ontario MWTPs (1 thousand tonnes in 1996) is similar to that from the Atlantic Provinces (0.9 thousand tonnes) and British Columbia (1 thousand tonnes), despite the population of Ontario being five times larger than that of the Atlantic region and three times larger than British Columbia. These similar loads are due to the fact that most of the population in Ontario is served by tertiary treatment compared to no treatment or primary treatment for coastal populations. Overall, total P release in 1996 from municipal wastewater was 4.3 thousand tonnes to inland waters, 0.44 thousand tonnes to Pacific coastal waters, 0.84 thousand tonnes to Atlantic coastal waters, and 0.002 thousand tonnes to Arctic coastal waters.

Total N released to lakes, rivers and coastal waters in Canada was approximately 80 thousand tonnes in 1996. This value represents a 17% increase over 1983 and a 7% increase over 1991 loads. Loads from Ontario and Québec were the largest of all provinces, because of their large populations. Overall, total N release in 1996 from municipal wastewater was 71 thousand tonnes to inland waters, 4 thousand tonnes to Pacific coastal waters and 5 thousand tonnes to Atlantic coastal waters.

The nutrient loads given above are for total N and total P. Only a portion of the total load is present in forms that have the potential to cause eutrophication or, in the case of N, have the potential to be toxic. For P, about 65 to 100% of the total load from sewage is bioavailable (Sonzogni et al. 1982, Tchobanoglous and Burton 1991, Berge and Källqvist 1998). For N, about 60% of the total load is in the form of free ammonia with the remainder being organic N (Tchobanoglous and Burton 1991).

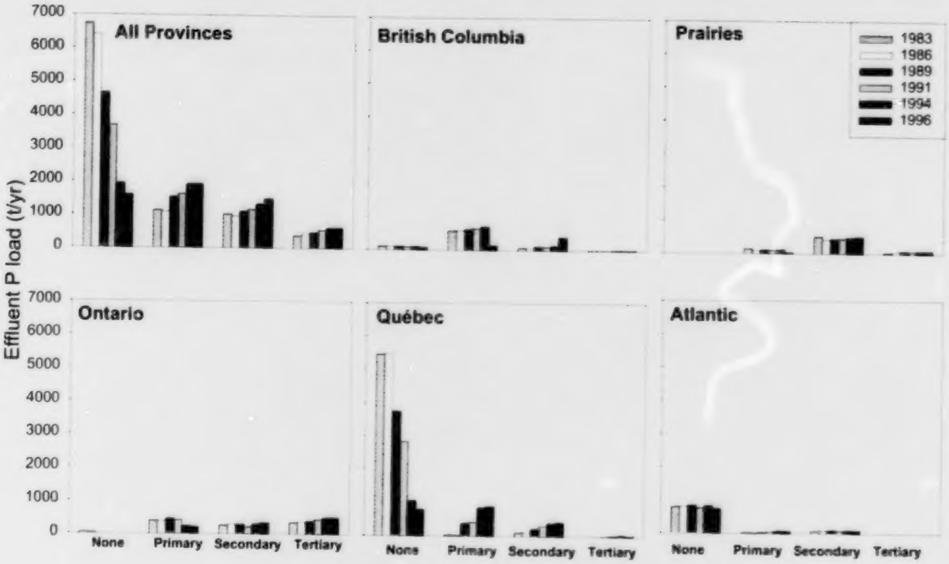


Figure 3.5. P loading as a result of municipal effluent releases for each provincial region of Canada, 1983-1996. Population served by primary, secondary or tertiary treatment (Environment Canada 1996a) multiplied by P removal values in Table 3.1. The population not served by sewage treatment was estimated as the difference between census population estimates for the region (Statistics Canada 1998a) and the Environment Canada (1996a) database of populations served by sewage treatment. Territorial data not presented, as the small numbers are not distinguishable at this scale.

Sources of nutrients in municipal wastewater include human waste, household cleaning products (laundry detergent, automatic dishwashing detergent and general-purpose cleaners), and by-products from industries that dispose wastewater to municipal sewer systems. Nitrogen enters domestic wastewater primarily from human waste; this source constitutes >90% of the household N load, with the dominant form of N being ammonia followed by organic N forms. Industrial discharges to the municipal sewer system also add N, however human waste generally represents most of the input.

Phosphorus sources to municipal wastewater are more varied than N sources. Prior to regulation of the P content of laundry detergent under Part III of the *Canada Water Act* (which restricted P content to8.7% by weight in 1970 and then to 2.2% in 1973), P loads from human waste and laundry detergent were roughly equal. Analysis of 1996 data shows human waste was the largest contributor to municipal P loading in Canada followed by commercial and industrial sources (Table 3.2). Automatic dishwashing detergents, general-purpose cleaners and laundry detergent each contributed ≤ 7% of the municipal P load.

In addition to routine operational wastewater discharge, MWTPs on occasion experience a discharge of raw sewage known as a bypass. A bypass differs from a combined sewer overflow in that raw sewage is released from a MWTP rather than from a sewer. Bypasses in both sanitary and combined sewers may occur for a number of reasons including significant increase in wastewater volume due to a storm event or spring thaw, operation problems such as equipment breakdown, or population and industrial growth exceeding the design capacity of the MWTP. In most provinces, bypasses are not allowed except in emergency situations, such as protecting basements from flooding, preventing damage to

Table 3.2. Sources of phosphorus in Canadian municipal wastewaters, 1996.

Source	P Content	Quantity	P Load (t/yr)	% of Total
Human waste	1.8 g/capita/day[1]	28,846,761[2] (population)	18,952	53
Laundry detergents	0% for 95% of detergent sold; 2.2% for 5% of detergent sold[3]	$150x10^3$ t[4]	165	<1
Automatic dishwashing detergents	6.0%[5]	$42x10^3$ t[6]	2,520	7
General purpose cleaners	2.2%[7]	$54x10^3$ t[8]	1,188	3
Commercial & industrial sources			12,763[9]	36
Household total	3.38g/capita/d[10]	28,846,761[2] (population)	**35,588**	100
Municipal load after treatment and discharge from MWTP[11]			**5,563**	

[1] Values typically range from 1.5 to 2.0 g/capita/day (Alexander and Stevens 1976; Balmer and Hultman 1988; Holtan et al. 1988; Barr Engineering 1993)

[2] Statistics Canada (1988a)

[3] The major manufacturers of laundry detergent (Proctor & Gamble, Lever Brothers) have eliminated P from laundry detergents sold in North America. Other manufacturers have not changed their formulations but they represent only 5% of the market (K.J. Ott, Albright & Wilson Americas Ltd., Mississauga, Ontario, personal communication).

[4] 1993 data (Environment Canada 1993)

[5] K.J. Ott, Albright & Wilson Americas Ltd., Mississauga, Ontario, personal communication

[6] In 1993 about $35x10^3$ t of dishwasher detergent used in Canada. The number of Canadian households with an automatic dishwasher increased by 29% from 4.4 million in 1992 to 5.7 million in 1997 (Statistics Canada 1992, 1997c). Given that dishwasher numbers increased on average by 5% per year between 1992 and 1997, dishwater detergent use likely also increased by 5% per year. Therefore in 1996, dishwater detergent consumption in Canada was likely $42x10^3$ t

[7] Grenon (1994)

[8] 1993 data (Environment Canada 1993c)

[9] Commercial and industrial P load calculated as the difference between the household total and the loads from human waste, detergents and cleaners

[10] Calculated from a 1991 Ontario study of municipal wastewater discharges (OMEE 1993)

[11] Total municipal load for 1996 from Figure 3.5. The municipal P load discharged to surface waters (i.e., 5.6 thousand t/yr) is less than the household total because 63 to 93% of the P is removed at the MWTP, depending on the level of sewage treatment provided.

MWTP equipment, or averting the wash-out of solids. Data from Ontario for 1991 showed that 75 of the 204 reporting MWTPs (or 37%) had bypasses (OMEE 1993), most of which occurred during the spring thaw months of March and April. The total annual volume was 2.2 million cubic metres for primary bypass and 9.6 million cubic metres for secondary bypass, representing 0.11 and 0.46%, respectively, of the total effluent volume treated in 1991 (OMEE 1993).

Sewage Sludge

Sewage sludge refers to the organic and inorganic solids resulting from decomposition and settling of wastewater as it undergoes treatment (Warman 1997). Sludge is produced during every stage of wastewater treatment (primary, secondary and tertiary, as well as in lagoons). Biosolids are the portion of the sewage sludge that has been stabilized through digestion to meet public health regulations for application to land (OMEE & OMAFRA 1996; WEF 1998). Biosolids are a semi-liquid material containing at least 10% suspended solids (WEF 1998). Solids produced during preliminary screening of sewage are generally not used as biosolids as they contain the large lumps, sticks, rags, pieces of metal and rocks that enter the wastewater treatment plant. They do not meet guidelines for application to agricultural land and, hence, are landfilled or incinerated (WEF 1998; M. Webber, wastewater consultant, personal communication).

The quantity of sludge generated increases as the level of sewage treatment increases. A primary treatment plant produces approximately 80 g sludge solids/person•day and contains a high percentage of grit and other inert material (Black et al. 1984). A secondary treatment plant yields about 115 g sludge solids/person•day whereas a tertiary treatment plant typically produces about 145 g sludge solids/person•day (Black et al. 1984). Sludge is digested either aerobically or anaerobically to kill pathogens and reduce water content.

As biosolids are a by-product of sewage treatment, they are rich in inorganic and organic materials and plant nutrients. Nutrient values for biosolids are typically 4% N and 2.5% P (Webber and Bates 1997). Biosolids are therefore a desirable additive to agricultural land because they recycle plant nutrients and organic matter to soil (Webber and Bates 1997). However, depending upon industrial contribution to the MWTP and its treatment process, biosolids may have high concentrations of heavy metals and pathogens (Webber and Bates 1997). Most provinces have guidelines for the management of land application of biosolids (Table 3.3; NSE 1992; CBCL Ltd. 1996; NBE 1996; OMEE & OMAFRA 1996; AEP 1997; BCELP 1998a; SERM 1998;) designed to match biosolid nutrient content with the nutrient demands of the crop, while limiting accumulation of micro-nutrients such as molybdenum, copper and zinc which can be toxic when present in high concentrations. In Canada, the application rates for digested sludge range from 8 to 25 tonnes dry solids/ha over two to five years depending on soil texture, field slope and soil pH (Table 3.3). In general, application rates are based on laboratory measurements of soil N content and can be further constrained by soil P concentrations.

With increasing landfill fees, agricultural application is becoming the most economical sludge management option. Sewage sludge production for Canada was estimated to be 500 000 t/yr (dry weight) in the early 1980s (OECD 1995). More recent data for the entire country do not exist, although the City of Toronto sewage sludge production for 1999 was about 70 000 t/yr (dry weight). Data from the early 1980s indicated that 42% of Canadian sludge was applied to agricultural land, 18% was landfilled and the remaining 40% was incinerated (OECD 1995). Based on sludge production of 500 000 t/yr (dry weight) and typical N and P contents of 4% and 2.5%, respectively, on a dry weight basis (Webber and Bates 1997) sewage sludge in Canada accounts for 20 thousand tonnes N and 12.5 thousand tonnes P per year. Of this total, 3.6 thousand tonnes N and 2.3 thousand tonnes P are landfilled; 8.4 thousand tonnes N and 5.3 thousand tonnes P are applied to agricultural land; and 8 thousand tonnes N and 5 thousand tonnes P are incinerated.

Discharges from Municipal Sewers

In urban areas, snowmelt or stormwater runoff from house roofs, parking lots, and streets empty into the municipal sewer system. Sewer systems built before the early 1940s disposed of household wastewater simply by discharging it directly into rivers, lakes or coastal waters. These sewer lines also carried stormwater and, hence, are referred to as combined sewer systems. As the deleterious consequences of raw sewage discharge became apparent, trunk sewers were built to divert the household sewage in combined sewers to sewage treatment plants. However, the cost of constructing treatment facilities with sufficient capacity to handle most of the stormwater flow was considered

Table 3.3. Nutrient derived guidelines for land application of biosolids for Canadian provinces (na indicates information not available).

Province	Sewage Biosolids	Application Guideline	Notes	Reference
Nova Scotia New Brunswick Prince Edward Island	Stabilized sludge only	5.6 t dry solids /ha (=160 kg N/ha/ 2 yr)	Only applied every 2 years due to upper limits of acceptable P concentration in soil Soil pH ≥ 6.0; optimal is 6.0-6.8 Field slope ≤ 8% Approval required before application allowed in NS	NSE 1992; CBCL 1996; NBE 1996
Québec		na	Heavy metal content of biosolids; soil P concentration	MEFQ 1997
Ontario	Aerobically digested sludge Anaerobically digested sludge	8 t solid/ha/ 5 yr 135 kg N/ha/ 5 yr	Soil concentration of P <60 mg/g in top 15 cm Soil pH ≥ 6.0 which may be achieved through liming	OMEE & OMAFRA 1996
Manitoba		15 t dry solid/ha/yr	Plant-available N cannot exceed 100 kg/ha/yr pH ≥ 6.0 Field slope ≤ 5% Sodium bicarbonate extractable P cannot exceed 60 µg P/g soil in the upper 15 cm	M. Van Den Bosch, Manitoba Conservation, personal communication
Saskatchewan	Undigested sewage	na	Not allowed on sandy soils Field slope <9%	SERM 1998
Alberta	Digested sludge Undigested sludge	25 t dry solids/ha/ 3 yr 5 t solids/ha/ 3 yr	Soil pH ≥ 6.5 which may be achieved through liming Medium to fine textured soils preferred Field slope < 9% with closed drainage	AEP 1997
British Columbia		na	organic matter must be <15% dry weight TKN must be <6% by weight C:N >15:1	BCELP 1998a

prohibitive. Consequently, the outfalls in the combined sewer system were left to act as "relief valves", and diversion or regulator structures were installed to divert flows in excess of sewer or MWTP capacity to these outfalls. As a result, whenever rain or snowmelt exceeds the design capacity of the sewer system or the MWTP, raw sewage and stormwater are released from the combined sewer outfalls into the receiving water. These discharges are referred to as combined sewer overflows (CSOs) and are typical for older parts of many Canadian cities.

Table 3.4. Nitrogen (TN, NO_3^-, NH_4^+) and total phosphorus (TP) loads from stormwater runoff and combined sewer overflows (CSOs) for selected Canadian urban areas expressed based on the size of the urban land (kg/ha) as well as the total load (t/yr).

City	Nitrogen		TP		Reference
	kg/ha	t/yr	kg/ha	t/yr	
Stormwater					
North York, ON	13 TN		2.6		Singer 1977
Burlington, ON					
Malvern (residential)	11.2 TN		1.3		Marsalek 1984
Aldershot (commercial)	14.3 TN		1.6		Marsalek 1984
Windsor, ON	4.6 TN		0.7		Singer 1977
Calgary, AB	10.3 NO_3; 0.16 NH_4^+	18 NO_3; 0.28 NH_4^+	0.22	0.4	Dixon 1994
Fraser River Basin					
Upper Fraser	1.3 TN; 0.5 NO_3; 0.1 NH_4^+	78 TN; 31NO_3; 7 NH_4^+	0.26	16	Stanley Associates Engineering Ltd. 1992
Middle Fraser	3.9 TN; 1.6 NO_3; 0.3 NH_4^+	13 TN; 5 NO_3; 1NH_4^+	0.78	3	
Thompson	2.0 TN; 2.0 NO_3; 0.2 NH_4^+	24 TN; 9 NO_3; 2 NH_4^+	0.39	5	
Lower Fraser	13 TN; 5.0 NO_3; 1.1 NH_4^+	764 TN; 306 NO_3; 66 NH_4^+	2.5	153	
Greater Vancouver				21	Environment Canada and BCELP 1992
CSO					
Greater Vancouver		213 TN; 88 NH_4^+		42	Environment Canada and BCELP 1992

In most urban areas developed since the 1950s, wastewater is collected in a separate sewer system consisting of sanitary sewers that transport household, commercial, institutional and industrial wastewaters, and storm sewers that convey stormwater. Surface runoff enters the storm sewer system through curb-side drains or other connections and is either discharged directly to the receiving water or passed through stormwater management facilities that may reduce flows and improve water quality (Marsalek and Kok 1997).

Detailed data are not available on the proportion of the Canadian population served by storm sewers versus combined sewer systems. However, most urban areas developed prior to the early 1940s are served by combined sewer systems. In large cities, many combined sewer outfalls exist and contribute to a great spatial extent of CSO impacts on a single receiving waterbody. For example, in Greater Vancouver, 252 stormwater and 53 combined-sewer outfalls discharge to the Lower Fraser River and estuary (UMA 1994, 1995). Winnipeg and Metropolitan Toronto have 75 and 79 combined-sewer outfalls, respectively, of which 74 of the Toronto outfalls are designated as priorities for pollution abatement action (Thorp et al. 1997; Winnipeg Web 1999). Given that combined sewers were used mostly in the older sections of cities where populations are now declining and that sewer separation programs have been undertaken by some communities during the past 25 years, the current Canadian population served by combined sewers is likely smaller than in 1969. All provinces now require that new sewer construction should be of the 'separate' type and municipalities are encouraged to replace combined sewer systems as soon as possible. CSO remediation is currently underway or planned in many cities, notably Vancouver, Edmonton, Winnipeg, Hamilton, Toronto, Ottawa, Montreal and Québec City.

Table 3.5. Concentrations of total phosphorus and total nitrogen in municipal wastewater.

Wastewater Type	Total Phosphorus (mg P/L)	Total Nitrogen (mg N/L)
Stormwater	0.33	2.18
Combined Sewer Overflows (CSO)	4	11

Data Sources: CSO data USEPA 1974
Stormwater data USEPA 1983

Quantities of both stormwater and CSOs vary temporally and with location, and depend on local climate, sewer design and drainage practices. Large-scale estimates gave an average annual stormwater discharge of about 760 L/capita/d for the Great Lakes basin (Marsalek and Schroeter 1988). However, the stormwater discharges would be 2000 to 3000 L/capita/d if the averaging was done just for the wet-weather days. The average annual discharge was about 473 L/capita/d for urban runoff and CSOs in the Greater Vancouver Regional District (Environment Canada and BCELP 1992). These flows greatly exceed the average municipal sewage flow of 300 L/capita/d. Expressed as an annual nutrient load, these discharges range from 1.4 to 14 kg/ha urban land for N and 0.2 to 2.6 kg/ha urban land for P (Table 3.4). A useful rule-of-thumb from a USEPA initiative, which assessed stormwater loadings from 22 locations (the Nationwide Urban Run-off Program, USEPA 1983), is that for a given urban population, stormwater will contribute about one-tenth the P load of a properly maintained secondary sewage treatment plant.

CSO or stormwater discharges are not routinely monitored in Canada; therefore, estimates of their volume and impact on receiving waters are rare. In general, stormwater volume is comparable in volume to the 4 300 million cubic metres discharged annually by MWTPs, however, its release is confined to wet weather periods (J. Marsalek, DOE, personal communication). Stormwater is also more dilute with respect to nutrients than raw or treated sewage or CSO. To estimate stormwater nutrient loads, stormwater volume (estimated to be 4 300 million m^3) was multiplied by 0.33 mg/L total P or 2.18 mg/L total N (Table 3.5) to give 1.4 thousand tonnes P and 9.4 thousand tonnes N per year. In contrast to stormwater, CSOs represent approximately 5% of MWTP annual discharge and are generally more dilute than raw sewage but higher in nutrient concentration than stormwater or treated sewage (Table 3.5). CSO nutrient loading was estimated by multiplying 5% of the annual MWTP discharge (4 300 million m^3) by 4 mg/L total P or 11 mg/L total N to give 0.9 thousand tonnes total P and 2.4 thousand tonnes total N (Table 3.5). Compared to sewage where 65 to 100% of the total P load is bioavailable and about 60% of the total N load is in the form of free ammonia (Sonzogni et al. 1982; Tchobanoglous and Burton 1991; Berge and Källqvist 1998), stormwater P is largely particulate (about 95%) and N is primarily in the form of organic material and nitrate+nitrite (Waller and Hart 1986; J. Marsalek, DOE, personal communication).

Septic Disposal Systems

A septic system is a small wastewater treatment system designed to treat wastewater from one or several homes. The system consists of a tank usually connected to a series of pipes set into trenches. Wastewater from the home flows into the tank. Heavier solids, or sludge, collect on the bottom of the tank; lighter solids, like hair and grease, float to the top and form a scum layer on the water. Both the top and bottom layers of solids are trapped in the tank where they are broken down by micro-organisms. The clarified wastewater is transported to a drain field by a pipeline. The drain field (sometimes called a tile field or leach field) consists of a series of distribution pipes set in trenches filled with gravel. These distribution pipes have holes so that the wastewater from the septic tank flows out into the gravel trenches then trickles through the gravel and into the soil under the drain field.

Approximately 25% of Canadians are served by septic disposal systems. These systems were originally designed for houses widely separated from their nearest neighbour, such as farmhouses and the occasional rural residence. Today, in many parts of the country, septic disposal systems present in too great numbers for the land base, not properly maintained, or built in unsuitable sites (i.e., too close to shorelines, in unsuitable rock or soil conditions) are sources of contamination to ground water and, ultimately, surface water. Wastewater from poorly maintained systems can also reach streams and lakes through overland flow. An average of 61% of septic field systems in various surveys of cottage systems in Ontario were not properly designed, constructed or maintained (Dillon et al. 1986).

The degree to which nutrients are retained in a septic field depends on the age of the field and soil characteristics, including adsorption capacity, natural drainage and permeability. Nitrate is not significantly retained by most soils (Bohn et al. 1985) and is readily transported by soil water and ground water. Phosphate migration in the saturated zone appears to be controlled primarily by sorption processes that significantly retard migration (Robertson 1995; Harman et al. 1996). However, as the adsorption capacity becomes saturated, migration of phosphate in the ground water zone may occur: low but perceptible rates of phosphate movement have been observed at several sites (Robertson 1995).

In general, septic tanks retain about 20 to 55% of the N and 25 to 40% of the P entering the system (Ryding and Rast 1989). Additional P is also retained in the unsaturated soil zone under the drain field so that, for properly designed septic tile filter beds, much of the P entering the system is retained either in the tank or in the unsaturated zone (see studies reviewed by Dillon et al. 1986; Robertson et al. 1998). Based on a population of 7 975 664 living in communities with a population less than 1 000 in 1996, an effluent volume of 160 L/capita/day (Siegrist et al. 1976), a P influent concentration of 15 mg/L (Laak 1974, Whelan and Titmanis 1982, Bicki et al. 1984), and a P retention coefficient of 72% for both the septic tank and drain field (an average of 10 sites studied by Robertson et al. 1998), 1.9 thousand tonnes of P is released from septic systems annually. Similarly, N release is 15.4 thousand tonnes/yr based on an influent concentration of 55 mg N/L (Epp 1984) and retention coefficient of 40% (Ryding and Rast 1989) for the same population.

Residential Solid Waste

Solid waste refers to the garbage collected from households and industries that requires incineration or disposal in a landfill. An estimated 10.5 million tonnes of residential solid waste were collected by municipalities across Canada in 1992 (Environment Canada 1996c). This averaged 0.38 tonnes per person per year. A 1989 Ontario survey showed that approximately one-third of residential solid waste was paper (of which half was newspaper), another third was organic material (food waste and yard waste), and the remaining third was a mixture of glass, metal (food containers and appliances), plastic, textiles and other materials (OME 1991). The majority of residential solid waste is landfilled, with the remainder being incinerated, recycled or composted. For example, in 1994, 75% of residential solid waste in British Columbia was landfilled or incinerated; the remainder was recycled (BCELP 1998b). Residential recycling is increasing. For example, only 0.5 million households in Ontario recycled in 1987 but by 1997, this number had increased to 3.1 million households (3Rs Information Partnership 1999).

The contribution from incineration of solid waste (residential and industrial) to atmospheric P release was estimated to be 123 tonnes in 1978 for Canada (Environment Canada 1983). Based upon 1995 data, incineration of municipal waste (solid and sewage sludge) also released about 293 tonnes of N as NO_x (Environment Canada 1999a). There are no national estimates of nutrient losses from landfills. Recent information from British Columbia indicates municipal landfills in the Fraser River basin leaked 145 tonnes of ammonia-N per year to surface waters in 1995 (Gartner Lee Ltd. 1997).

The Composting Council of Canada estimates that approximately 20% of the 7 million tonnes of organic waste produced annually is composted (CCC 1998). Compost is defined as the solid mature product resulting from composting, a managed process of bio-oxidation of solid heterogeneous organic substrate including a thermophilic phase (CCC 1998). In 1997, materials composted included leaf and yard waste, wood, animal manure, industrial/commercial/institutional waste (including supermarket and restaurant waste), paper materials (sludge, boxboard and unrecyclable paper), residential food waste, and fish waste and other marine materials (Antler 1997). Composting is conducted at centralized composting sites as well as in backyard composters. Currently, approximately 1.2 million backyard composters have been distributed to Canadian households (CCC 1998). Compost is a beneficial additive to soils as it supplies organic matter and nutrients, improves water retention and reduces soil compaction. High quality compost from centralized composting sites is sold for use in agriculture, horticulture, landscaping and home gardening. Medium quality compost is used for erosion control and roadside landscaping. Low quality compost is used as a landfill cover or in land reclamation projects (CCC 1998). Since 1991, composting has gained a strong foothold in the Canadian waste management infrastructure. Approximately 600 thousand tonnes of finished compost was produced in 1996 (Antler 1997). Typically, vegetable compost contains 2-7.9 kg/t N (mean = 5.0 kg/t N), and 6.2-11.9 kg/t P (mean = 9.1 kg/t P; Gagnon et al. 1999). Assuming that all of the finished compost is applied to land, 3 thousand tonnes N/yr and 5.5 thousand tonnes P/yr are applied to soil as compost.

3.2 Industrial Discharges

Industries produce waste that may be released directly to the atmosphere, land, surface waters or ground water or, in the case of solid or liquid waste, transferred off-site for land-based application, disposal in landfills or treatment in municipal wastewater plants. Most light industries discharge their wastewater into the municipal sewage system, their solid waste to municipal landfills, and their gaseous waste to the atmosphere. Industrial waste discharged to a licensed facility (e.g., a municipal wastewater treatment plant) is not typically regulated by the provincial or federal government with respect to the quantity or quality of their discharge. Some municipalities have sewer-use bylaws regulating the strength of industrial discharges to sewer systems. MWTPs are designed to reduce bacterial contamination and biochemical oxygen demand, remove most P (in the case of facilities with advanced P removal) and, in some cases, convert ammonia to nitrate. There is little ability to do more than volatilize or dilute many industrial wastes.

Large industries (e.g., pulp mills, mining operations, large manufacturing plants, etc.) that independently discharge their waste to air, water or land must obtain operating permits from the provincial government in which they are situated or, in the case of industries operating in the Territories, by the Federal Government (unless this responsibility has been devolved to the Territory). This permit may stipulate the quantity and/or quality of waste that can be disposed to land, air or water.

Table 3.6. Number of manufacturing industries in Canada, number of industries with discharge permits, number of permitted industries reporting wastewater discharge, and number of permitted industries with wastewater discharge that report total P, nitrate-N (NO_3-N) or ammonium-N (NH_4^+-N).

Region	Number of Manufacturing Industries[1]	Number of Industries with Discharge Permits[2]	Number of Permitted Industries reporting Nutrients in Wastewater Discharge[3]	Number of Permitted Industries with Wastewater Discharge[3] Reporting:		
				TP	NO_3^-	NH_4^+
Atlantic	1,919	283[a]	6	1	0	3
Québec	10,603	370	101	42	38	34
Ontario	14,471	274	160	100	28	59
Prairies	4,827	175[b]	49	36	19	31
BC	4,378	805	16	9	3	14
Territories	41	223	4	3	3	2
Canada	36,236	2,130	336	191	91	142

[1] Statistics Canada 1997d

[2] A permit to discharge does not necessarily mean that the industry is discharging directly to the environment.

[3] Data not supplied by New Brunswick, Prince Edward Island and Nova Scotia. Québec data are only for industries discharging to the St. Lawrence River basin.

[a] New Brunswick supplied the number of water approvals. A plant can have more than one water approval.

[b] Alberta supplied the number of major industries required to report the quality of their discharge on a monthly basis. Alberta has many more industries with approvals with different frequency of discharge or discharge to municipal systems.

Wastewater Discharges

Estimates of wastewater N and P loading were obtained from the provincial and territorial governments for industries with operating permits. The data spanned the period 1988 to 1999. New Brunswick, Prince Edward Island and Nova Scotia did not supply data; Québec data are only for industries that discharge to the St. Lawrence River basin. Of the data obtained from the provincial and territorial governments, 336 industries (16% of all industries with permits to operate) reported discharge of nutrients in wastewater (Table 3.6). Not all industries are required to measure N and P loads even though they had a wastewater discharge. Therefore, industrial N and P loads are an underestimate of the quantity actually being discharged. Of the 336 industries in Canada that reported nutrients in wastewater discharges, total P concentration was measured by 191 industries, NO_3^- by 91 industries and NH_4^+ by 142 industries. Not all industries reported total N or the forms of N that would allow calculation of total N.

Industrial N and P loads to surface waters are presented in Figure 3.6. These values represent only the measured and reported amounts and likely underestimate actual loads. Comparison of N and P loads among regions and among sectors cannot be made because not all provinces provided data and because provincial/ territorial governments differ in the parameters they require a particular industrial sector to monitor. Given these caveats, the available data indicate that at least 2 048 tonnes per year of total P, 7 588 tonnes per year of nitrate-N (NO_3-N), and 4 231 tonnes per year of ammonium-N (NH_4^+-N) are discharged from permitted industries in Canada to surface waters (Figure 3.6).

The data in Table 3.7 provide an illustration of the discrepancies found in industrial records. Although total values for ammonia loading are similar for both the National Pollutant Release Inventory (NPRI) and provincial/territorial records, the loads differ considerably between certain industrial sectors. NPRI

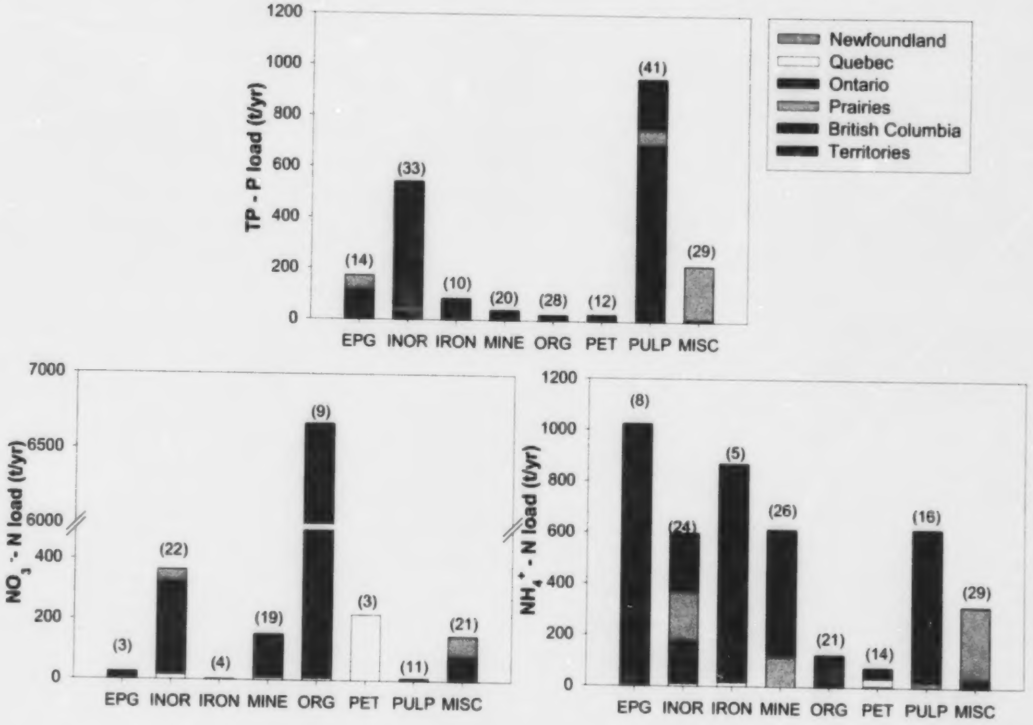


Figure 3.6. Industrial wastewater nutrient loads (total P, nitrate and ammonium) by region and sector, as provided by the agencies listed below. Numbers above each bar indicate the number of reporting industries by sector. Industrial sectors are abbreviated as follows: EPG: Electric Power Generation; INOR: Inorganic Chemicals; IRON: Iron and Steel; MINE: Metal Mining and Refining; ORG: Organic Chemical Manufacturing; PET: Petroleum Refining; PULP: Pulp and Paper; MISC: Miscellaneous, includes metal casting, food processing, industrial minerals, and metal, plastic fabricating and finishing. Data sources: Newfoundland loads were calculated from 1997 self-reported data provided by the Newfoundland Office of Environmental Protection of Environment Canada; Québec loads were calculated for 1990 to 1997 from data obtained from St. Lawrence Vision 2000 (1998) and represent 106 industries discharging into the St. Lawrence River; Ontario loads were calculated from 1995 MISA data provided by Ontario Ministry of Environment; 1999 Manitoba loads were provided by Manitoba Environment; Saskatchewan loads were calculated from 1988 to 1991 data provided by Saskatchewan Environment and Resource Management; Alberta loads were calculated from 1997 data provided by Alberta Environmental Protection; British Columbia loads were calculated for 1994 to 1998 using data provided by the British Columbia Ministry of Environment, Lands and Parks; Northwest Territories and Nunavut loads were calculated from 1997 data for the Northwest Territories provided by Environment Canada; Yukon loads were calculated from 1998 data provided by INAC and Environment Canada; data were not available for NB, NS and PEI.

Table 3.7. Comparison of total ammonia in industrial wastewater reported to the National Pollutant Release Inventory (NRPI) for 1996 (Environment Canada 1996b) and ammonium (NH_4^+) load reported to provincial/territorial departments. (Provincial/territorial data sources as for Figure 3.6.)

Industrial Sector	NPRI data (t (NH_4^+ + NH_3) /yr)	Provincial/Territorial data[1] (t NH_4^+ /yr)
Metal casting		1
Electric power generation	71	1024
Food processing	528	75
Inorganic chemical manufacture	315	595
Iron and steel	868	867
Metal and plastic fabricating and finishing	22	3
Metal mining and refining	737	613
Miscellaneous	2	238
Organic chemical manufacturing	143	123
Petroleum refining	72	75
Pulp and paper	1220	616
Total	3978	4231

[1] Data sources as for Figure 3.6.

requires all Canadian industries employing a workforce equivalent to 10 or more full-time employees and manufacturing or using ammonia (or any one of another 175 substances in 1996 or 276 substances in 2000) in quantities greater than 10 tonnes to report these releases. Exempted facilities include those engaged in research, testing or education, and growing, harvesting, or extraction of natural resources. Ammonia loads provided to NPRI may be reported as ammonium (NH_4^+), ammonia (NH_3), or ammonium or ammonia expressed as N. The NPRI total ammonia loads are not necessarily based on measured concentrations but may be based on engineering estimates, mass balances or emission factors. In contrast, provincial/territorial data are based on measured concentrations and report ammonium or ammonia expressed as N. However, not all industries are required to report their ammonia or ammonium loads to the provincial or territorial government.

In addition, NPRI indicates that 6 421 tonnes of N as ammonia were injected underground in 1996 (Environment Canada 1996b). This process involves injecting wastes into known geological formations, generally at great depths, and occurred largely in Alberta and, to lesser extent, in Saskatchewan. It was associated mainly with fertilizer manufacturing, petroleum refining and organic chemical manufacturing sector in Alberta and the mining industry in Saskatchewan.

Air Emissions

Industries release N to the atmosphere as nitrous oxide (N_2O), other nitrogen oxides (NO_x) and ammonia (NH_3).

Nitrous oxide (N_2O) is largely produced from the combustion of coal, petroleum products and natural gas. Of the total Canadian emissions for 1995 (98 thousand tonnes N_2O-N), industrial combustion-related emissions accounted for only 3 thousand tonnes of N (Table 3.8). Another 2 tonnes of N were emitted to the atmosphere as a result of manufacturing nitric acid, an intermediate product formed during the manufacture of N fertilizers. Twenty-two tonnes of N were also emitted during production of adipic acid, a chemical used to manufacture nylon and polyurethane foam. However, the sole

Table 3.8. Emissions of nitrogen compounds to the atmosphere from industrial sources. N_2O data from Environment Canada 1997a; NO_x data from Environment Canada 1999a; 1995 ammonia data from Vézina 1997.

Industrial Sector	1995 N_2O (10^3 t N)	1995 NO_x (10^3 t N)	1995 NH_3 (10^3 t N)
Combustion-related emissions			
Power and steam generation	2	77	0.4
Industrial	1	9	
Industrial Processes			
Chemical manufacture	24	7	8.8
Metal mining and refining		8	
Metal & plastic fabricating & finishing		0.1	
Pulp and paper manufacture		17	
Iron and steel manufacture		8	
Petroleum/petrochemical industry		113	8.5
Miscellaneous		36	9.6
Canadian Industrial Total	27	275	26.9
Canadian Anthropogenic Total	98[1]	750[2]	623[3]

[1] includes 38x10^3 t N from agriculture in 1996 (Desjardins and Keng 1999), 31x10^3 t N from transportation combustion, and 2x10^3 t N from other sources

[2] includes 393x10^3 t N from transportation combustion, 64x10^3 t N from forest fires, and 18x10^3 t N from other sources

[3] anthropogenic total for 1995 includes 570x10^3 t N from agriculture, 2.4x10^3 t N from non-industrial fuel combustion, 4.2x10^3 t N from transportation combustion, and 18.9 x10^3 t N from other sources

Canadian producer introduced new technology in to reduce N_2O emissions (Environment Canada 1997a).

Nitric oxide and nitrogen dioxide (i.e., NO_x) are also produced during the combustion process. Canadian NO_x emissions from anthropogenic sources for 1995 were 750 thousand tonnes N, of which 52% was attributable to transportation (i.e., cars, trucks, airlines and boats) (Environment Canada 1999a). The petroleum/petrochemical industry accounted for 113 thousand tonnes of N, representing the second highest source of anthropogenic NO_x emissions (Table 3.8).

Ammonia is released to the atmosphere by chemical and chemical products industries. According to Environment Canada (1999b) after Vézina (1997a), 27 thousand tonnes of N as ammonia were released to the atmosphere by Canadian industries in 1995. Of this, 33% was produced by industries engaged in the manufacture of chemicals (Table 3.8).

A 1978 study of atmospheric P emissions in Canada estimated that 1.98 thousand tonnes of P were released by industrial processes, most of which was associated with the production of fertilizer (Environment Canada 1983, Table 3.9). Fertilizer production entails grinding of phosphate rock and production of phosphoric acid, and accounted for 0.5 thousand tonnes of atmospheric P emission in 1978.

Solid Waste Disposal

About 10 million tonnes of solid waste were generated by Canadian industries in 1995 (Statistics Canada 1998b). A 1989 study showed that solid waste generated by Ontario industries was composed of paper products (23%), wood (21%), other organic material (11%), metal material (11%), glass (5%),

Table 3.9. Total phosphorus emissions to the atmosphere from industrial sources - 1978. Data source: Environment Canada 1983.

Industrial Sector	Phosphorus (t)
Phosphate rock processing	58
Phosphoric acid production	688
Phosphate fertilizer production	488
Elemental phosphorus production	682
Other chemical industries	19
Primary aluminum production	30
Miscellaneous sources	15
Canadian Industrial Total	1980
Canadian Total[1]	3402

[1] includes 783 t from fertilizer application, 255 t from forest fires, 199 t from fuel combustion for stationary sources, 123 t from solid waste incineration, 47 t from slash burning and 15 t from transportation sources.

plastic (3%), tires (2%) and other miscellaneous material (24%) (OME 1991). Major contributors to industrial solid wastes in Ontario for 1989 were the construction industry (30% of total tonnage) and the communications and services sector (22% of total tonnage). Large quantities of the solid waste generated by industry are potentially recyclable (e.g., paper products, wood, plastic, metal, tires). Since the 1989 Ontario study considerable effort has been directed at incorporating "reduce, reuse, recycle" activities in industrial solid waste management efforts.

Of the approximately 10 million tonnes of industrial solid waste generated in Canada in 1995, 78% was landfilled 20% was recycled, and the remainder incinerated (Statistics Canada 1998b). The contribution from incineration of solid waste (residential and industrial) to atmospheric P release was estimated to be 123 tonnes in 1978 for Canada (Environment Canada 1983). Incineration of industrial and commercial waste also released about 226 tonnes of NO_x-N in 1995 (Environment Canada 1999a). There are no national estimates of nutrient losses from landfills. A recent British Columbia study reported that ammonia-N loading from pulp and paper mill landfills in the Fraser River basin was 6 tonnes per year (Gartner Lee Ltd. 1997).

3.3 Agricultural Loading

Plant nutrients (in the form of fertilizer and manure) are essential additions to agricultural lands as they contribute to maintenance of soil health and optimal crop yield. However, application of fertilizer or manure in excess of plant requirements can lead to a build-up of nutrients in the soil and, once the soil's retention capacity is surpassed, their eventual loss to the environment. This loss can be to surface or ground waters or to the atmosphere. Conversely, underuse of fertilizers can lead to a loss of soil fertility (known as mining of the soil) and will eventually result in crop reductions. Effective use of nutrients contributes to agricultural sustainability by enhancing farm profitability through efficient use of costly soil additives and minimization of environmental impacts due to excessive nutrients.

Agriculture is practised widely in seven of Canada's 15 ecozones (see Figure 1.1 for ecozone locations). The Pacific Maritime ecozone has some of the mildest and wettest climatic conditions in Canada. Farmland is confined to the Fraser River valley and eastern coastal area of southern Vancouver Island and consists of dairy, hog, and poultry operations as well as speciality products such

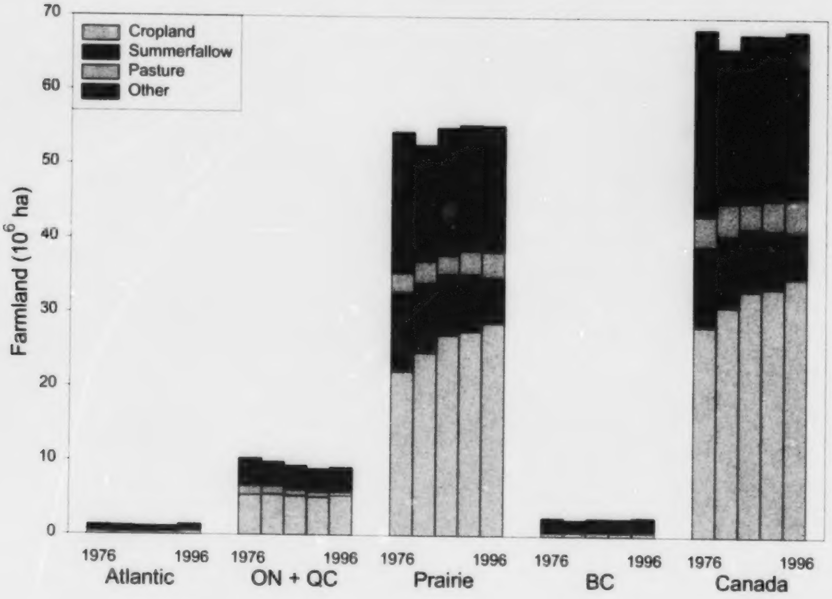


Figure 3.7. Use of Canada's agricultural land for the census years 1976-1996. The category 'Other' includes farmland area occupied by farm buildings, lanes, woodlots, wetlands and tree windbreaks. Data are from Statistics Canada (1997b).

as nursery, floriculture, greenhouse vegetables, and berry crops. In the Montane Cordillera ecozone, tree fruit production and viticulture dominate under the mild climate of the semi-arid valleys in the south whereas beef cattle production is common in the northern valleys and high-elevation plateaux. The Boreal Plains supports productive agriculture across a band located just north of the Prairies ecozone and is dominated by the production of cereals, oilseeds and forages. About two-thirds of all farmland in Canada is located in the Prairies ecozone. Here, grain and oilseed production are the major farming activities, followed by beef cattle production. The Mixedwood Plains ecozone is characterized by a warm, humid climate conducive to the production of a wide range of products including most of Canada's dairy products, vegetables and speciality crops. Pockets of mixed agriculture also extend north of the Mixedwood Plains into the Boreal Shield. In the Atlantic Maritime ecozone, cool-season vegetables, forage, and dairy production are the major outputs. Agriculture is the dominant land use throughout Prince Edward Island and is concentrated in river valleys elsewhere in the Atlantic Maritime ecozone.

Approximately 7% of Canada's total land area, 68 million hectares, is agricultural land of which 46 million hectares is cropland, pasture, or summerfallow (Figure 3.7; Statistics Canada 1997b). The total area of farmland in Canada has remained relatively constant for the last 50 years. However, regional changes have occurred in recent decades: for example, there have been large increases in cropland area in the Prairies, lesser increases in British Columbia, and decreases in Ontario, Québec and the Atlantic Provinces (Figure 3.7). In addition, the number of farms has declined, from 280 043 in 1991 to 276 548 in 1996, while the size of individual farms has increased (Statistics Canada 1997a).

Fertilizer Production and Use

Commercial fertilizers are a major source of nutrients to crops. Improved crop varieties developed over the last 40 years have increased productivity substantially, with yields doubled for many crops. These new hybrid crops take up more nutrients from the soil than their lower-yielding predecessors. For example, an average of 200 thousand tonnes of N and 36 thousand tonnes of P was applied annually to the western Canadian grain crop between 1883 and 1953 (Flaten and Hedlin 1988). By comparison, more than 1.1 million tonnes of N and 170 thousand tonnes of P were applied to the western Canadian grain crop in 1986, a four-fold increase over the 1883 to 1953 average. During the same period (1953 to 1986), the Prairies cropland base increased by 1.6 times from 18 to 28 million ha. Natural nutrient supplies are insufficient to reach high yield potentials, making it necessary to use supplementary fertilizer nutrients.

Both N and P fertilizers are manufactured in Canada. Nitrogen fertilizers are based on ammonia, which can be used directly as a fertilizer or converted to solid forms (urea, ammonium phosphate, ammonium nitrate, and/or ammonium sulfate) or to N solutions (e.g., anhydrous ammonia, urea-amonium nitrate solution). Canada produced 4 025 thousand tonnes of N fertilizer in 1996 (FAO 1999). Of this, approximately 60% was exported (mostly to the United States); an additional 210 thousand tonnes of N as fertilizer were imported into Canada (FAO 1999). There were 1 576 thousand tonnes of N as fertilizer applied to crops in Canada in 1996 (Agriculture and Agri-Food Canada 1998).

Phosphate fertilizers are produced from phosphate rock and sulfuric acid. Canada imports all its phosphate rock from Togo or the United States: 1 140 thousand tonnes in 1996 (Paul Lansbergen, Canadian Fertilizer Institute, personal communication). Phosphate rock is 12 to 18% P (Potash and Phosphate Institute 1988). About 175 thousand tonnes of P as fertilizer were produced in 1996 in Canada (FAO 1999) and of this, 7.9 thousand tonnes of P were exported; an additional 129 thousand tonnes of P as fertilizer were imported. In 1996, 297 thousand tonnes of P as fertilizer were applied to croplands in Canada (Agriculture and Agri-Food Canada 1998).

Data on the land base to which fertilizer is applied and the quantity of fertilizer used by each province are available from Statistics Canada (1997a) and Agriculture and Agri-Food Canada (1998a), respectively (Appendix 1). Based on these data, average P and N application in 1996 ranged from 10 kg P/ha and 60 kg N/ha in the Prairies to 33 kg P/ha and 86 kg N/ha in the Atlantic Provinces (Figure 3.8). Comparisons from 1981 to 1996 showed that fertilized land increase by about 9% in the Atlantic Region, declined by about 10% in Québec, 5% in Ontario, and 0.2% in British Columbia, and increased by 32% in the Prairie Provinces. Over the same time period, fertilizer N inputs (per ha) increased for the Atlantic and Prairie regions and Québec, declined in Ontario, and remained relatively constant in British Columbia. Fertilizer P inputs (per ha) have remained relatively constant in Québec and the Atlantic region, but declined in British Columbia, Ontario and the Prairie Provinces over the same time. Overall, 297 thousand tonnes of P as fertilizer and 1 576 thousand tonnes of N as fertilizer were applied to 24 943 thousand ha of Canadian cropland in 1996.

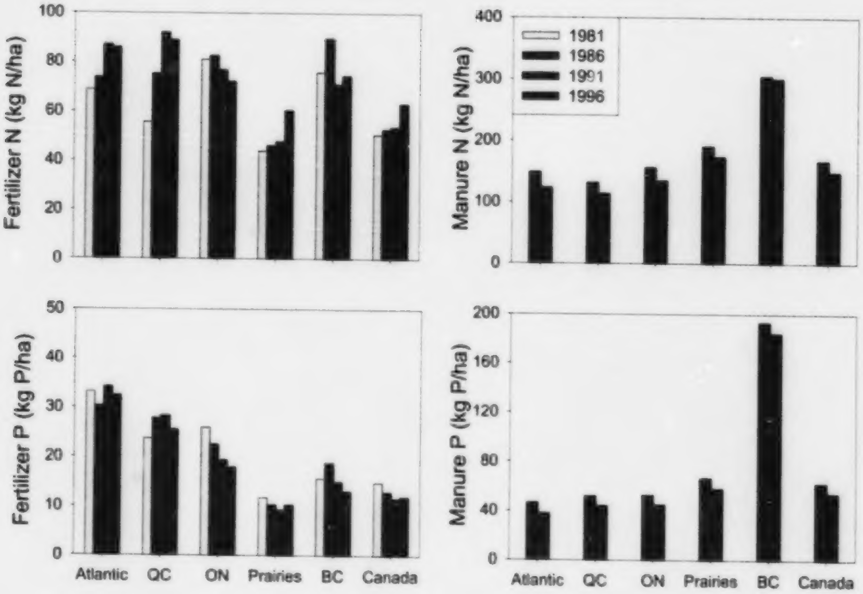


Figure 3.8. N and P added as fertilizer (1983-1996) and manure (1991-1996) to Canada's cultivated land. Fertilizer data are from Statistics Canada (1997a) and Agriculture and Agri-Food Canada (1998a) for census years. Manure data are from Statistics Canada (1997a) and Table 3.10.

Manure Production and Use

Manure consists of undigested feed (liquid and solid), metabolic waste, bedding material, and wasted feed and water. In 1996, 4 680 585 beef cattle, 1 227 732 dairy cattle, 102 255 149 poultry and 14 224 664 other livestock were reared in Canada (Statistics Canada 1997a). A total of 97 920 farms or 44% of all farms in Canada in 1995 were considered to be "livestock farms" with an additional 20% of all farms reporting that some livestock were kept, but were not their main agricultural activity (Agriculture and Agri-Food Canada 1997). These statistics indicate that manure storage and disposal is presently a management consideration on many Canadian farms. Historically, manure was not a problem because most farms had a mix of livestock and crops and the small livestock numbers on an individual farm compared to many farms today meant it was feasible to recycle nutrients between crops and livestock.

Almost all manure produced on Canadian farms is applied to agricultural land (Patni 1991). Some of the manure is excreted directly to pasture and is not practically accessible. However, manure produced when livestock are housed or confined in collecting yards is readily available for use as fertilizer on fields. Manure is an important additive to agricultural soils because it supplies both nutrients and organic matter. Depending upon the type of livestock and the rations being fed, fresh manure can contain 50 to 80% of the N and P originally present in the feed. Not all nutrients in manure are immediately available to crops, however. Some are tied up in organic forms and become available over time as the material decomposes, thereby acting as a nutrient source over several years. In addition, the organic matter in manure contributes to improved soil structure, nutrient and water-holding capacity, and drainage.

Table 3.10. Total nitrogen and phosphorus manure production (t/yr) calculated by multiplying the number of animals in each province (Statistics Canada 1997b) by the N and P content of manure for each province (from Appendix 1).

Year	Atlantic	QC	ON	Prairies	BC
Nitrogen (10^3 t/yr)					
1976	23.0	106.8	178.7	209.8	33.2
1981	24.5	119.4	189.2	201.8	41.4
1986	24.7	104.0	165.9	198.0	35.3
1991	23.0	102.2	154.0	217.1	36.8
1996	22.4	109.4	151.0	259.7	39.5
Phosphorus (10^3 t/yr)					
1976	4.3	28.2	38.2	52.5	12.2
1981	4.7	33.2	41.1	48.2	15.9
1986	4.9	28.6	36.3	46.2	14.7
1991	4.6	27.9	33.9	50.9	15.1
1996	4.5	29.7	33.1	56.3	15.7

Estimates of manure N and P production (tonnes/year) for each province were derived from data on livestock numbers (Statistics Canada 1997b) and manure N and P production per animal type (Appendix 2; Table 3.10). Nutrients are lost from manure between the time it is produced and its application to fields. Much is in the liquid portion of manure, and significant losses occur if this liquid is allowed to run off feedlots and from manure storage facilities. In addition, a large proportion of manure N is in the form of ammonium (NH_4^+) and, upon exposure to air, can be converted to ammonia gas (NH_3) and lost to the atmosphere. To estimate manure application to farmland, we assumed a 40% loss of N from cattle manure and a 25% loss of N from swine and poultry manure from the time of excretion to field application (Environment Canada 2000b; MacDonald 2000a). Phosphorus does not exchange with the atmosphere and we therefore assumed zero loss between production and field application. Beef cows are commonly left out to pasture for most of the year; consequently, we did not include beef cows in our calculations of manure nutrient loads to cropland.

Based on manure production estimates, stated assumptions as to nutrient loss, and area of application reported in the agricultural census (Statistics Canada 1997a), average manure P and N application was estimated for each province (Appendix 3). Average manure application in 1996 ranged from 114 kg N/ha (Québec) to 301 kg N/ha (British Columbia) and 38 kg P /ha (Atlantic) to 184 kg P/ha (British Columbia) (Figure 3.8). The land base to which manure was applied increased in the Prairie Provinces (by 27%), Ontario/Québec (by 19%), the Atlantic Provinces (by 19%), and in British Columbia (by 9%) from 1991 to 1996, coinciding with decreased manure application rates (kg N or P per ha) across all regions. Over the same time period, cattle numbers across the whole country increased by 14%, and pigs and poultry numbers each increased by 8%. Compared to commercial fertilizer, manure was applied at much higher rates: a total of 139 thousand tonnes P and 482 thousand tonnes N was applied to 2 579 thousand hectares of Canadian cropland in 1996.

Other N Inputs

Nitrogen fixed from the air by legume crops can add to the soil's N supply. The ability of legumes to improve soil N concentrations is the basis for crop rotation schedules on many farms. Atmospheric N_2 fixation rates for legume species range from 53 kg N/ha/yr for chickpeas to 100 kg N/ha/yr for clover (Appendix 4; F. Selles and R. Lemke, Agriculture and Agri-Food Canada, personal communication).

These estimates were multiplied by the area planted in legumes in 1996 (Statistics Canada 1997a) to give legume inputs of N to soil. N input from atmospheric N_2 fixation by legumes ranged from 20 thousand tonnes N in the Atlantic region to 477 thousand tonnes N in the Prairies, for a total of 773 thousand tonnes N fixed by legumes in Canada in 1996 (Table 3.11).

Harvest of the legume crop removed 896 thousand tonnes of N in 1996 (calculated by multiplying legume production [Statistics Canada 1997a] by the concentration of N in the harvested portion [Appendix 4; F. Selles and R. Lemke, Agriculture and Agri-Food Canada, personal communication]). Legumes acquire their N by absorbing it from the soil through their root system and by symbiotic fixation of atmospheric N_2. However, for the legume, it is more energy efficient to assimilate soil N than to fix N_2 from the atmosphere; thus, only 5 to 65% of N in the legume is fixed from the atmosphere; the remainder is absorbed from the soil (Biederbeck et al. 1996). The result of this is that harvest of legumes often removes more N than was fixed from the atmosphere. Incorporation of legume residues (the dead non-harvested portion of the plant) into the soil increases soil N reserves. A small fraction of this N, 10 to 30%, may become available to the next crop (Gleig and MacDonald 1998). The remaining N in the legume residue is incorporated into the soil organic matter and will contribute to the N supplying capacity of the soil.

In addition to N fixed by legumes, deposition from the atmosphere is a significant N source. In Canada, atmospheric deposition in the form of nitrate and ammonium (dissolved inorganic N, DIN) has steadily increased since the 1900s and, in the 1990s was estimated to supply, on average, 3.44 kg N/ha/yr east of the Manitoba-Ontario border compared to 0.80 kg N/ha/yr west of this border (see Section 3.6). Given 34 918 733 ha were in cropland in 1996, atmospheric loading of DIN was a source of approximately 43 thousand tonnes of N. National data are not available to estimate loadings expressed as total N.

Table 3.11. Nitrogen inputs due to legume atmospheric nitrogen fixation in 1996.

Region	Legume N fixation (10^3 t)
Atlantic	19.5
Québec	90.1
Ontario	150.8
Prairie	476.5
British Columbia	36.1
Total	773.0

Nutrient Uptake by Agricultural Crops

The amount of N and P removed from the field as harvested crop can be calculated from data on crop yields and the nutrient content of the harvested crop (Beauchamp and Voroney 1994; Bolinder et al. 1997). The harvested crop is the grain for crops such as wheat, barley and oats, hay for alfalfa and clover, and the fruit for fruit crops. Straw is assumed to be returned to the soil, either directly or in manure. In 1996, a total of 2 491 thousand tonnes of N and 386 thousand tonnes of P were removed in the harvested crop for Canada (Figure 3.9).

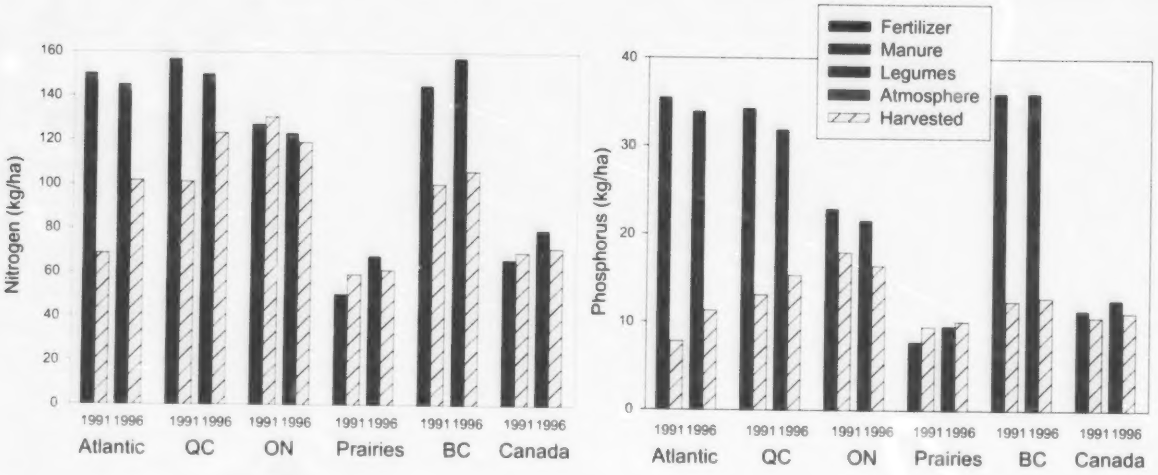


Figure 3.9. The amount of total N and total P added per hectare to agricultural fields as chemical fertilizer, manure, legume N-fixation and atmospheric deposition (dissolved inorganic N, only, for atmospheric deposition), and the amount removed in harvest in each region of Canada, 1991 and 1996.

Nutrient balances for Canada in 1996 indicate that crop removal of N and P was 89% and 87%, respectively, of nutrient additions as manure, fertilizer and, in the case of N, atmospheric deposition and legume fixation (Figure 3.9). In 1996, an estimated 2 784 thousand tonnes of N were added to agricultural soils (1 576 thousand tonnes from fertilizer, 384 thousand tonnes from manure, 773 fixed by legumes, 8 thousand tonnes as biosolids, and 43 thousand tonnes as atmospheric deposition). Likewise, an estimated 442 thousand tonnes of P were added to agricultural soils (297 thousand tonnes from fertilizer, 139 thousand tonnes from manure, and 5 thousand tonnes as biosolids). Although all provinces have surpluses of N, British Columbia and the Atlantic region tended to have the largest. In the case of P, cropland in the Prairie Provinces was under fertilized relative to the harvest (Figure 3.9).

Nutrient Losses from Agricultural Activity

Based on our assessment of N inputs to Canadian cropland (from fertilizer, manure, biological N fixation, atmospheric deposition, and sewage biosolids) and outputs as a result of crop harvest, the N surplus for Canada in 1996 was 4.3 kg/ha for the total area of agricultural land (68 million hectares) or 8.4 kg/ha for cropland (35 million hectares). Similarly, the P surplus in 1996 was 0.8 kg/ha for all agricultural land or 1.6 kg/ha for cropland. Our N estimate is comparable to values produced by the Organization for Economic Co-operation and Development (OECD) which estimated the N surplus in Canada to have been 13 kg/ha in 1995-1997 and 6 kg/ha in 1985-1987 based upon the total area of agricultural land (OECD 2000). These surpluses are among the lowest of the 28 OECD countries studied. OECD data are not available for P.

Although national data on nutrient surpluses or deficits are useful for examining patterns among countries, it should be recognized that these are average values and that, within a country, regional N surpluses or deficits may occur. For example, a recent analysis of residual N in Canadian farmland reported values of ≥ 41 kg/ha in the lower Fraser Valley of British Columbia; the corridor of agricultural land from Lethbridge through Red Deer to Edmonton in Alberta; the Melfort area in northeastern

Table 3.12. Proportion of farmland at risk of causing nitrogen contamination of water leaving farmland through overland flow (runoff) or leaching for regions in Canada where the soils have a water surplus, 1996 (MacDonald 2000b).

Province	Total Farmland[1] Area (1000 ha)	Farmland[1] area where soils have a water surplus (1000 ha)	Farmlands[1], where soils have a water surplus, classed according to their risk for nitrogen contamination of runoff or seepage water (%)		
			Low (0-6 mg/L N)	Medium (6.1 - 14 mg/L N)	High (> 14 mg/L N)
British Columbia	2018	70	6	25	69
Ontario	4577	4200	39	44	17
Québec	2267	1900	58	35	6
Atlantic	575	400	82	15	3

[1] Farmland refers to the sum of all *Agricultural Profile of Canada* land classes except 'All Other Land'. (Statistics Canada 1997a).

Saskatchewan; the Red River Valley in Manitoba; southwestern Ontario, the area around Lake Simcoe and the lower Ottawa Valley in Ontario; the St. Lawrence Lowlands in Québec and the region south of Québec City; the Annapolis Valley in Nova Scotia; and the Saint John River Valley in New Brunswick (MacDonald 2000a). However, most farmland in the four western provinces (69-92%) and the Atlantic Provinces (85%) had residual N levels of < 41 kg/ha in 1996. MacDonald (2000a) also reported residual N increased between 1981 and 1996 for all provinces except British Columbia. The percentage of farmland showing an increase of ≥ 5 kg/ha residual N ranged from 27% in British Columbia to 80% in Manitoba.

In establishing a reference level against which to monitor and assess changes in N surpluses, some studies suggest that values greater than 100 kg/ha annual should be taken as a baseline for increasing risk to possible nitrate leaching in ground and surface waters (Schleef and Kleinhanss 1994). A study by the USDA (1997) categorizes nutrient balances as high if the nutrient input (from fertilizer, manure and N fixed by legumes) exceeds output by more than 25%, moderate if nutrient input exceeds output by less than 25%, and negative if input was less than output. Canada's recent N surplus qualifies as low compared to the 100 kg/ha baseline suggested by Schleef and Kleinhanss (1994). Inputs (2784 thousand tonnes N and 442 thousand tonnes P) exceeded outputs (2491 thousand tonnes N and 386 thousand tonnes P) by 11% for N and 13% for P, indicative of moderate inputs for both N and P on the USDA scale.

Losses to Surface or Ground Waters from Agricultural Fields

National estimates of the quantity of N and P lost to surface and ground waters from agricultural lands are not available for Canada. In the USA, losses from cropland and from pasture and rangeland represented 39 and 13 %, respectively, of the 8 158 thousand tonnes of N and 31 and 17 %, respectively, of the 2 015 thousand tonnes of P discharged annually to surface waters (Carpenter et al. 1998). A recent assessment of N losses from Canadian agricultural fields related the average N surplus to the estimated volume of water leaving the fields through overland flow (runoff) or leaching for regions of Canada where the agricultural soils have a water surplus (i.e., Pacific Maritime, BC; Boreal Shield of northern Ontario, Québec and Manitoba; Mixedwood Plains of southern Ontario; and Atlantic Provinces; Macdonald 2000b). In British Columbia, 5% of the total agricultural land has a water surplus and 69% of this area was predicted to generate runoff or seepage water with N concentrations

> 14 mg N/L. In Eastern Canada, 17% of Ontario, 6% of Québec and 3% of Atlantic farmland that has soils with a water surplus was predicted to produce runoff or seepage water with > 14 mg N/L (Table 3.12). Risk of P loss from Canadian agricultural fields (based on the approach used for N) has only been completed for Québec and only identifies relative risk based on farm practices and soil, topographic, weather and other environmental conditions (Bolinder et al. 2000).

Although national estimates of the quantity of N and P lost to surface and ground waters from agricultural lands are not available for Canada, watershed or regional assessments have been attempted by assigning nutrient loss coefficients to different types of land (e.g., pasture, cropland, coniferous forest, etc.) or by estimating nutrient losses from agricultural land as percent of the nutrients applied in fertilizer and manure (Bolinder et al. 2000). Attempts to assess the portion of surplus nutrients moving to surface and ground waters have also employed field-scale measurements of losses from runoff, infiltration and leaching, and tile drains (MacDonald 2000b). For example, agricultural sources are estimated to contribute 70% of the N and 75% of the P load discharged from the Yamaska River, Québec to the St. Lawrence River (Chambers et al. 2000; see also Yamaska River case study in Section 4.3).

The information on nutrient surpluses/deficits and nutrient losses to surface and ground waters are for total N and total P. Only a portion of the total load is present in forms with the potential to cause eutrophication or, in the case of N, to be toxic. N and P are typically lost from agricultural lands in particulate forms that are not readily available to plants. However, recent research undertaken in Alberta indicated that 65-100% of the total P exported from agricultural watersheds was in dissolved forms (Anderson et al. 1998, Cooke and Prepas 1998). The predominance of dissolved P forms in runoff may be unique to the Boreal Plains and caused by watershed characteristics such as low-sloping topography (resulting in longer leaching times) and a high proportion of organic soils. Because dissolved forms are more-readily available for plant growth, they are likely to cause more rapid biological responses than particulate nutrients.

Losses to Surface or Ground Waters from Livestock Housing & Handling

A 1995 survey of Canadian farms found that 60% or 133 655 farms reported they stored manure. Of these, about 11% (14 885 farms) stored liquid manure, including all dairy farms most hog farms and about 25% of all poultry and egg farms. Of farms reporting liquid manure storage, the two most common methods were unlined lagoons (33%) and open tanks (31%). More than 95% of the farms storing liquid manure in 1995 reported that storage sites were at least 15 m from the nearest watercourse and at least 30 m from any well used for domestic purposes (Agriculture and Agri-Food Canada 1997). The exceptions were 595 farms in southern Ontario, southern Québec and the Atlantic Provinces that stored liquid manure within 15 m of a watercourse, and 565 farms in the Prairies Provinces, southern Ontario, southern Québec and Atlantic Provinces that stored liquid manure within 30 m of a well.

In addition to liquid manure storage, 126 470 farms stored solid manure in 1995 (Agriculture and Agri-Food Canada 1997). Farms with liquid manure storage also stored solid manure. All beef farms and mixed livestock farms and most poultry and egg operations and dairy farms reported storage of solid manure. Of the farms storing solid manure, 60% stored it in an open pile without a roof. Approximately 99% of farm operators reported storing solid manure storage more than 15 m from the nearest watercourse, and 97% stored it more than 30 m from any well used for domestic purposes (Agriculture and Agri-Food Canada 1997).

Other sources of agricultural nutrients to surface waters are discharges such as milk house wastes draining through connected tile drains to streams, cattle allowed access to streams, erosion of stream banks due to livestock trampling or tillage, and greenhouse waste. There are few regional and no national data on nutrient losses from livestock housing and handling facilities or from manure storage facilities to surface or ground waters. Localized studies have typically shown a 40% loss of N from cattle manure and a 25% loss of N from swine and poultry manure to soils, water and air (OMAFRA 1997; Environment Canada 2000b). For example, a study of agricultural nutrient management in the Lower Fraser Valley, British Columbia estimated that for housed beef and dairy cattle, approximately 15 to 20% of the manure N content was lost to the air; manure storage resulted in another 10 to 20% loss to air and 0 to 10% loss in runoff or infiltration (Brisbin 1995). In the same study, N losses for hog manure were estimated at 15 to 20% to the air when the animals were housed and 10 to 20% (most to the air) during manure storage.

National estimates were not calculated for N and P losses to surface or ground water from livestock housing and handling facilities or from manure storage facilities.

Losses to the Atmosphere

Within the agricultural sector, the majority of N losses to the atmosphere occur as a result of the volatilization of ammonia from livestock manure and fertilizers, and through the production of nitrous oxide by bacterially-mediated denitrification of soil nitrate. There is also some ammonia loss from the decomposition of crops, as well as losses of nitric and nitrous oxide from the soil following mineralization of organic matter. These losses of N from manure and fertilizer can represent a potentially serious economic loss to the farmer and impact to the environment. Losses also occur as a result of fuel combustion and biomass burning but these will not be addressed in this section. Wind and water erosion of soil are also transport pathways for nutrients. Agricultural management processes can accelerate natural rates of erosion (Podbury and Stushnoff 2000; Shelton et al. 2000); however, nutrient losses associated with erosion will not be addressed here.

Ammonia release to the atmosphere largely occurs as a result of storage and handling of livestock manure and fertilizer application. The majority of ammonia losses from manure and chemical fertilizers occurs during application to the soil surface and tends to be higher during warm, dry weather. Between 3 and 60% of applied NH_3 can be lost depending on the type of material applied, soil conditions (moisture and pH), and the length of time elapsing before the fertilizer is incorporated into the soil (McGinn and Janzen 1998). In 1995, Canadian agricultural emissions of N as ammonia were estimated to be 570 thousand tonnes (Vézina 1997). Of this number, manure accounted for 87% of the total ammonia N emissions with fertilizer making up the balance. Of the manure emissions, cattle manure accounted for 59%; poultry manure for 21%, and pig manure for 19% of the ammonia N.

Nitrous oxide (N_2O) release is also associated with fertilizer use. When either manure or inorganic N fertilizers are applied to soil, most of the N is oxidized to nitrates before being taken up by plants, resulting in a loss of nitrous oxide to the atmosphere. In Canada, nitrous oxide emissions from agricultural activity increased from 31 thousand tonnes N per year from 1981 to 1991 to 38 thousand tonnes of N in 1996 (Desjardins and Keng 1999). The largest agricultural source of nitrous oxide is

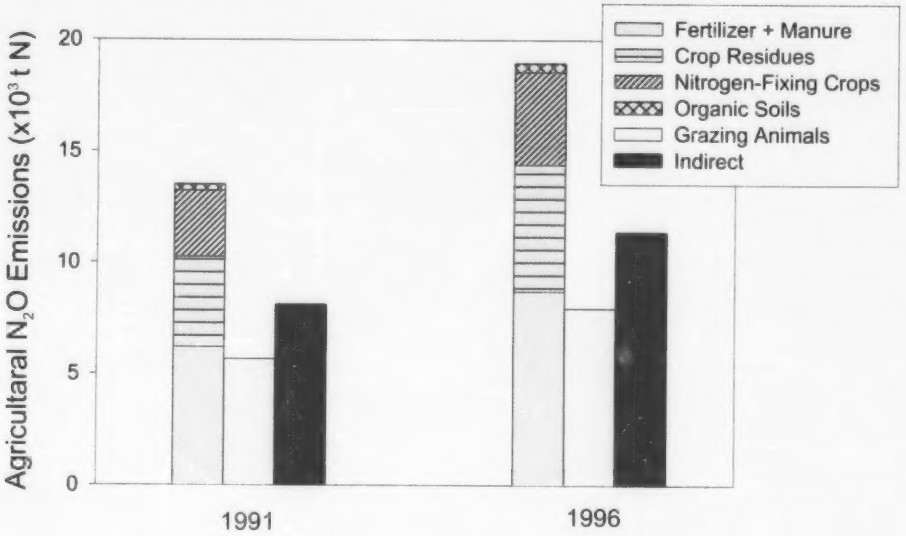


Figure 3.10. Canadian agricultural sources of nitrous oxide emissions for 1991 and 1996. Direct emissions occur from soils and grazing animals. Indirect emissions occur from N volatilization, leaching and runoff. Data adapted from Environment Canada 1997a, Monteverde et al. 1997, Desjardins and Keng 1999 and Desjardins and Reznek 2000.

direct soil emissions resulting largely from the application of manure and commercial fertilizer and which accounted for 6 thousand tonnes of N in 1991 and 9 thousand tonnes of N in 1996 (Figure 3.10; Desjardins and Keng 1999). Soil emissions can be subdivided into losses associated with manure and fertilizer use (23% of total emissions), crop residues (15% of total emissions), N-fixing crops (11% of total emissions), and cultivation of organic soils (<1% of total emissions) (Desjardins and Keng 1999). Grazing animals and manure management were responsible for 8 thousand tonnes of N in 1996 (21% of total N_2O emissions), while indirect emissions from the precipitation of NH_3 and NO_x and N leaching from soil were responsible for another 11 thousand tonnes of N (30% of total N_2O emissions) (Monteverde et al. 1997; Desjardins and Keng 1999).

Nitric oxide and nitrogen dioxide (i.e., NO_x) are also released to the atmosphere as a result of agricultural activities. Although national values for NO_x release from agriculture are not available, the quantity of N oxides released from soil (as a of product of nitrification and denitrification) may be similar to that from industrial sources (Janzen et al. 1998).

A small portion of P is lost to the atmosphere during fertilizer application. In 1978, an estimated 783 tonnes of P were released to the atmosphere as a result of fertilizer application, based on an emission factor of 9.81 x 10^{-4} tonnes P emitted per tonne fertilizer material used (Environment Canada 1983). Most of the lost P is in the form of phosphate.

3.4 Aquaculture and Fisheries Enhancement

In its broadest definition, aquaculture is "the cultivation of aquatic organisms (fish, molluscs, crustaceans, other invertebrates, unicellular algae, macroalgae, and higher plants) using extensive or intensive methods in order to increase the production or yield per unit area or unit volume to a level

Table 3.13. Fish and invertebrate species licensed for production in Canada. (Data from Moccia and Bevan 1996 and BC Fisheries 1998).

Fish

Lake sturgeon (*Acipenser fulvescens*)
Atlantic salmon (*Salmo salar*)
Brown trout (*Salmo trutta*)
Brook trout (*Salvelinus fontinalis*)
Lake trout (*Salvelinus namaycush*)
Arctic charr (*Salvelinus alpinus*)
Chinook salmon (*Oncorhynchus tshawytscha*)
Coho salmon (*Oncorhynchus kisutch*)
Pink salmon (*Oncorhynchus gorbuscha*)
Rainbow trout (*Oncorhynchus mykiss*)
Lake whitefish (*Coregonus clupeaformis*)
Lake herring (cisco) (*Coregonus artedii*)
Muskellunge (*Esox masquinongy*)
Northern Pike (*Esox lucius*)
Creek chub (*Semotilus atromaculatus*)
White sucker (*Catostomus commersoni*)
Bluntnose minnow (*Pimephales notatus*)
Fathead minnow (*Pimephales promelas*)
Redbelly dace (*Phoxinus eos*)
Atlantic halibut (*Hippoglossus hippoglossus*)
Atlantic cod (*Gadus morhua*)
Lumpfish (*Cyclopterus lumpus*)
Wolffish (*Anarhichas lupus*)
Finescale dace (*Phoxinus neogaeus*)
Common shiner (*Luxilus cornutus*)
Golden shiner (*Notemigonus crysoleucas*)
Emerald shiner (*Notropis atherinoides*)
Common carp (*Cyprinus carpio*)
Goldfish (*Carassius auratus*)
Brown bullhead (*Ameiurus nebulosus*)
Channel catfish (*Ictalurus punctatus*)
American eel (*Anguilla rostrata*)
Largemouth bass (*Micropterus salmoides*)
Smallmouth bass (*Micropterus dolomieu*)
Bluegill (*Lepomis macrochirus*)
Pumpkinseed (*Lepomis gibbosus*)
Black crappie (*Pomoxis nigromaculatus*)
Walleye (*Stizostedion vitreum*)
Sauger (*Stizostedion canadense*)
Yellow perch (*Perca flavescens*)
Tilapia (*Oreochromis, Sarotheradon, Tilapia*)
Haddock (*Melanogrammus aeglefins*)
Winter flounder (*Pleuronectes ferrugineus*)
Yellowtail flounder (*Pleuronectes americanus*)
Striped bass (*Morone saxatilis*)

Invertebrates

Pacific oyster (*Crassostrea gigas*)
Geoduck clam (*Panopea generosa*)
Manila clam (*Tapes philippinarum*)
Gallo mussel (*Mytilus galloprovincialis*)
Green sea urchin (*Strongylocentrotus droebachiensis*)
European oyster (*Ostrea edulis*)
Little neck clam (*Protothaca staminea*)
Japanese scallop (*Pecten yessoensis*)
Blue mussel (*Mytilus edulis*)
Crayfish (*Orconectes immunus, O. virilis, O. propinquus, Cambarus robustus, C. bartonii*)

above that obtained naturally in a particular aquatic environment" (Mariculture Committee of the International Council for the Exploitation of the Sea cited in Boghen 1989). The aquatic organisms resulting from these husbandry techniques are intended for use as food, ornamental or baitfish production, or for resource enhancement (Boghen 1989).

Aquaculture in Canada is currently a small industry. Its roots lie in hatcheries established to produce fingerlings for provincial stocking programs. Aquaculture is rapidly expanding, however, and now includes production of approximately 45 fish and eight invertebrate species (Table 3.13). Of these 40 fish species, rainbow trout (including steelhead), Atlantic salmon and arctic char represent the bulk of fish produced for human consumption (DFO 1998). Clams, oysters, mussels, and scallops compose the majority of invertebrate production in Canada (DFO 1998). In 1996, 53 thousand tonnes of finfish and 19 thousand tonnes of shellfish were harvested as a result of aquaculture in ten provinces representing a value of $350 million (Table 3.14). Of this production, 58% was in marine environments and 42% in freshwater operations (Table 3.15), the majority in British Columbia and New Brunswick.

Aquaculture operations range from small private ponds stocking dozens of fish for private capture to large ocean cage cultures producing thousands of tonnes of fish per year. These operations can be categorized using three broad categories: open, semi-closed and closed (Landau 1992). The

Table 3.14. 1996 Canadian aquaculture production statistics and value by province (Data Source: DFO 1998).

Province/Territory	Finfish Production (t)	Shellfish Production (t)	Value ($000)
Newfoundland	1,319	386	6,139
New Brunswick	16,380	733	123,818
Nova Scotia	1,511	773	10,421
Prince Edward Island	64	10,493	14,444
Québec	1,000	100	4,100
Ontario	4,000	0	16,060
Manitoba	(1)	0	(1)
Saskatchewan	775	0	3,420
Alberta	110	0	660
British Columbia	27,731	6,475	170,744
Canada	52,907	18,960	349,910

(1) Majority of aquaculture is private so no data are available

distinction among these depends on the water source. In open systems, a natural water source, such as the ocean or a lake, is used and enclosures such as clam beds or fish cages are installed within the water body (Landau 1992). Enclosures range from a retaining structure held in place with poles or stakes to floating rafts or cages. Semi-closed systems use natural water sources in specially designed facilities (Landau 1992). This system allows for greater control over growing conditions and results in more uniform growth and increased production per unit area. Hatcheries are typically semi-closed systems. Finally, closed systems involve no water exchange with the natural environment. Organisms are raised at very high densities in tanks or ponds in water extensively treated and recycled (Landau 1992). Nutrient release from fish production systems results from excretion of dissolved or solid waste and from unconsumed feed. Finfish operations require the introduction of feed, which increases the potential for fertilization of surrounding waters. In contrast, shellfish operations in Canada rely on natural food sources from the surrounding water (Bob Hooper, Memorial University, personal communication); there are no food-related nutrient additions to the environment.

Of the three categories of aquaculture operations, only open and semi-closed systems have water exchange with a natural water body (lake or coastal water) and, therefore, the potential to affect the

Table 3.15. Percentage of Canadian aquaculture sites situated in marine or freshwater environments for each province in 1996. Data Source: Provincial lease authorities compiled for Canadian Aquaculture Industry Alliance (CAIA 1998).

Province/Territory	Marine	Freshwater
Newfoundland	100	0
New Brunswick	63	37
Nova Scotia	100	0
Prince Edward Island	100	0
Québec	7	93
Ontario	0	100
Manitoba	0	100
Saskatchewan	0	100
Alberta	0	100
British Columbia	78	22
Canada	58	42

natural environment. Moreover, emissions and treatment of effluents from closed aquaculture operations are controlled by provincial legislation. In addition to the number of fish cultivated, the quantity and quality of the feed are the most important factors determining nutrient loss to the environment because they determine both feed wastage and excretion losses (Persson 1991; Cho and Bureau 1997). Food quantity depends on the fish species under cultivation as well as the production goals for the operation. The objective of the aquaculturist is to match food supply with consumption rates to maximize economic gain while minimizing possible negative impacts of poor water quality (resulting from overfeeding) on target organisms, some of which are sensitive to water quality (e.g., trout). Inherent in this goal is a critical balance between the proportion of the nutrients utilized for fish growth and that lost to the water as excess feed and/or excretion (Cho et al. 1994).

Considerable study has been undertaken in recent years to improve feed quality so as to increase biological conversion of food into animal production and reduce wastage (Cho et al. 1994; Cho and Bureau 1997). A detailed study by Cho et al. (1994) showed that P and N for common feeds ranged from 1.4 to 13.6% N and 0.3 to 5.9% P, depending upon the ingredients used in the formulation. As a result of improvements in feed formulation, N and P content has decreased from about 1.62% P and 8.45% N in the early 1980s to about 0.9% P and 7.2% N (Ackefors and Enell 1990). In an assessment of waste discharge from salmon aquaculture (DFO 1997) assumed the nutrient content of feed to be 1.2% P and 6.4% N.

Improvements in feed quality have reduced the feed coefficient (wet weight of feed used to wet weight of fish produced). Thus, the feed coefficient has declined at least 3.5-fold since the inception of salmon farming in Norway approximately 45 years ago and between 1.5 and 2-fold since 1985 in British Columbia (DFO 1997). The DFO study used feeding coefficients for farmed Atlantic and Pacific salmon in British Columbia of 1.15 and 1.3, respectively, giving an average feeding coefficient (weighted for the proportion of each species in cultivation in British Columbia) of 1.2.

The method of feeding can also influence the rate at which food is lost to the environment. In a study of salmon cages in Sooke Basin on southern Vancouver Island, Levings (1994) noted automatic fish feeders resulted in deposition of waste food and particulate matter at a rate of 54.3 g dry weight/m^2/d, which was significantly higher than deposition rates of 19.1 g dry weight/m^2/d observed when fish were hand fed.

Studies on the loading of nutrients from aquaculture operations into aquatic ecosystems have tended to focus on large salmonid cage-culture operations in coastal waters and, to a lesser extent, fresh waters. Studies of rainbow trout and Atlantic salmon aquaculture have shown nutrient losses (both dissolved and particulate) from cage cultures ranged from 3 to 35 kg P and 20 to 260 kg N per tonne fish (Penczak et al. 1982; Ackefors and Enell 1990; Holby and Hall 1991; Jensen 1991; Johnsen and Wandsvik 1991; Hall et al. 1992; Levings 1994; Einen et al. 1995; MacIsaac and Stockner 1995). These studies have typically shown only about 20 to 30% of nutrients added to the aquaculture operation are incorporated into fish biomass and removed at harvest; the other 70 to 80% of added nutrients is lost to the environment in the form of metabolic waste, faeces and uneaten food fragments.

In an assessment of nutrient loading from Swedish fish farms to coastal waters, Ackefors and Enell (1990) assumed P loss to be 9.5 kg/t fish produced (based on a feed coefficient of 1.5 and a feed P content of 0.9%) and N loss to be 78 kg/t fish produced (based on a feed coefficient of 1.5 and a feed N content of 7.2%). In a similar assessment for Atlantic and Pacific salmon operations in British Columbia, DFO (1997) data gave losses of 9.2 kg P and 43 kg N per tonne of fish produced (based on a feed coefficient of 1.2 and feed contents of 1.2% P and 6.4% N). Assuming the DFO (1997) N and P loss coefficients are applicable to the entire finfish industry in Canada, nutrient introduction from Canadian aquaculture operations is 204 t/yr P and 956 t/yr N to inland waters, and 282 t/yr P and 1 320 t/yr N to coastal waters (calculated by multiplying the total finfish production for Canada in Table 3.14 by the percent of that production that is marine or inland in Table 3.15 and then multiplying by 9.2 kg P/yr or 43 kg N/yr). By comparison, a population of 100 000 served by secondary sewage treatment releases 25 t/yr P and 365 t/yr N as wastewater (based on 3.38 g P/capita/d and 80% P removal, and 10 g N/capita/d).

In addition to commercial aquaculture operations, direct fertilization has been used to enhance the production of sport fish, especially anadromous sockeye salmon, in oligotrophic lakes and rivers in British Columbia. Nutrients are added in an attempt to enhance food supply (algal and, in turn, benthic invertebrate or zooplankton abundances) for salmon fry. For example, nutrient additions to the Keogh River, British Columbia to increase summer average nutrient concentrations from 25 µg N/L and <1µg P/L to 30 to100 µg N/L and 10 to 15 µg P/L caused a 5 to 10-fold increase in periphyton biomass resulting in a 1.4 to 2.0-fold increase in salmonid fry weights (Johnston et al. 1990). In Kootenay Lake, British Columbia, five years of N and P additions of 47 t P/yr and 206 t N/yr resulted in increased zooplankton densities that increased both spawner size and fecundity of Kokanee salmon stocks (Ashley et al. 1997). Overall, nutrient additions to a variety of British Columbia lakes and rivers have enhanced bacteria, phytoplankton and zooplankton abundances with an associated 60% increase in smolt weight of juvenile salmon, helping revitalize the British Columbia salmon fishery (Stockner and MacIsaac 1996).

3.5 Forest Management Practices

Forests, particularly those occupying steep terrain and supplied with abundant precipitation, contribute a large proportion of the water that enters streams and lakes. Forests efficiently cycle large quantities of N and other nutrients with very small losses to surface waters. Evidence from small catchment studies and broader-scale monitoring programs indicates that streams draining undisturbed forest generally have high water quality with low concentrations of dissolved nutrients and suspended sediments. Forest management practices that disrupt the cycle of nutrients between the soil and trees may increase stream water concentrations of dissolved N, base cations and to a lesser extent P. However, nutrient losses to stream water from forest management practices have been defined for relatively few sites in Canada. The effect of forest management practices on water quantity and quality is difficult to assess, therefore, and confounded by climatic, topographic and vegetation diversity across the country.

Harvesting

Manipulation of forest watersheds can drastically affect water quality of streams and lakes. The most significant effects are on the physical characteristics of the water. Sediment from logging roads, skid trails, and landings and soil erosion in steep terrain increase the concentration of suspended sediments (including particulate forms of N and P) in streams. Under high flows, the concentration of suspended solids in streams in recently clear-felled forest catchments can rise dramatically compared to baseflow conditions. For example, sediment transport in streams in a coastal British Columbia forest increased

Table 3.16. Effect of harvesting on stream nitrogen and phosphorus concentrations $^{+}$.

	NO_3-N (mg/L)		PO_4-P (µg/L)			
Province	Control Stream	Treated Stream	Control Stream	Treated Stream	Treatment	Reference
NB	0.12	0.60	--	--	100% clearcut	Krause (1982)
ON	0.03	0.02	10.0	20.0	100% clearcut	Nicolson et al. (1982)
ON	0.11	0.08	10.8	9.0	75% clearcut	Nicolson (1988)
BC	0.04	0.50	--	--	100% clearcut	Feller and Kimmins (1984)
BC	0.003	0.006	0.08	0.07	100% clearcut	Scrivener (1987)

$^{+}$ Maximum average annual concentration in post-treatment period.

up to 10-fold owing to stream bank erosion following clearcutting. Impacts vary with the soil type, slope, climate, and severity of management treatment. Degradation of water quality is most likely to occur where forest management practices are poorly implemented.

Clearcutting of forests has also been shown to increase the concentration of dissolved nutrients (including nitrate). Forest removal affects nutrient leaching from forests by eliminating nutrient uptake by trees for a period of time. The impact of clearcutting on nitrate concentrations in first-order streams draining small, undisturbed, forested catchments at the Hubbard Brook Watershed in New Hampshire, USA is well documented (control 0.2 mg/L NO_3-N, clearcut 3.9 mg/L NO_3-N; Hornbeck et al. 1987; control < 1mg/L NO_3, stripcut 9 mg/L NO_3, clearcut 30 mg/L NO_3 one year after harvest, Martin et al. 2000). Increases in stream nitrate concentrations after clearcutting have also been observed for some but not all catchment studies in Canada (Table 3.16). When increases in nitrate were observed they were quite low, relative to the Hubbard Brook results. Few studies have examined changes in dissolved phosphate concentrations and those that have, typically show concentrations either remained unchanged after logging or exhibited short-lived increases. For example, three lakes on the boreal shield of northern Ontario showed little change in water quality in the first three years after experimental logging of 45 to 75% of the watershed (Steedman 2000). An exception to this observation is a recent study on boreal shield streams in Québec that reported no increase in nitrate export from streams draining harvested as compared to reference watersheds but a two-fold increase in total P and total N export in the year following harvest (Lamontagne et al. 2000). Although export from these boreal streams decreased in the years following harvest, nutrient export was still greater for harvested than reference watersheds two (total P) or three (total N) years after harvest. Impacts of timber harvest on nutrient concentrations are magnified by increases in water production after clearcutting. For example, P (dissolved and suspended) leaching from a jack pine-black spruce watershed in northwestern Ontario increased from 0.07 to 0.22 kg/ha/yr immediately after clearcutting (Nicolson et al. 1982).

The impact of harvesting on stand or catchment nutrient retention can also be assessed by mass budget calculations. Nutrient balances have been computed for only a few forests (e.g., Table 3.17). Studies of nutrients removed in the harvest products and by post-harvest leaching of the soil at an infertile coniferous site and a moderately fertile deciduous site suggest that, over the long term, clearcut harvesting will produce only small net gains to or losses from soil nutrient pools, with the exception of N in the tolerant hardwood forest. As a consequence the overall impact, viewed from the perspective of a forest crop rotation of 60 to 100 years, will typically represent only small increases or decreases in nutrient inputs to aquatic systems. For example, a paleolimnological study of phytoplankton

Table 3.17. Nutrient balance for a coniferous and a deciduous forest in Ontario. Data source: Mahendrappa et al. 1986; N. Foster, Natural Resources Canada, unpublished data.

Forest Type	**Nutrient (kg/ha/yr)**	
	N	**P**
Boreal conifer forest (mature jack pine on podzolic soil)		
Input		
Precipitation	3.0	0.5
Dry deposition	0.7	0.1
Atmospheric nitrogen fixation	0.02	--
Outputs		
Harvest bole	1.3	0.1
Harvest crown	1.5	0.15
Stream flow	0.1	0.01
Gain/Loss to soil nutrient pool	+0.8	+0.3
Temperate tolerant hardwoods (mature sugar maple on podzolic soil)		
Input		
Precipitation	8.7	0.3
Dry deposition	2.2	0.1
Atmospheric nitrogen fixation	0.3	--
Outputs		
Harvest bole	1.8	0.1
Harvest crown	1.9	0.1
Stream flow	3.7	0.02
Gain/Loss to soil nutrient pool	+3.8	+0.2

composition did not reveal evidence of eutrophication following the onset of logging ca. 1870-1890 in the watershed of a northern Michigan lake (Scully et al. 2000). However, over the short-term (i.e, the first several years following harvest), comparatively small nutrient inputs may affect lakes, particularly those with naturally-low nutrient concentrations. For example, total P and total organic N concentrations were significantly higher in boreal shield lakes in Québec in the three years following timber harvest as compared with lakes in undisturbed watersheds (Carignan et al. 2000). In the lakes exposed to timber harvest, a significant increase in phytoplankton abundance was observed in the first year following harvest, an increase that likely would have been even greater if not for lower light penetration caused by inputs of dissolved organic carbon that coloured the water (Planas et al. 2000).

The impact of timber harvest on concentrations of dissolved solids (including N and P) depends on the intensity of harvest, forest cover, soil type, and slope steepness. Impacts will also vary with the level of protection afforded during disturbance, through the use of lake shore reserves and buffer strips. Increases in nitrate are dependent on atmospheric N deposition, nitrification rates in soil, and N demands of regenerating vegetation and stream biota. Recovery of water quality is particularly dependent on the rate of re-establishment of vegetative cover.

Site Preparation and Slashburning

Site preparation is commonly applied after logging to help provide a more favourable environment for the establishment and early growth of planted seedlings. Exposure of mineral soil by site preparation can result in increased surface runoff and off-site movement of nutrients in dissolved form or in sediment transport. The greatest negative impacts may be associated with complete removal and

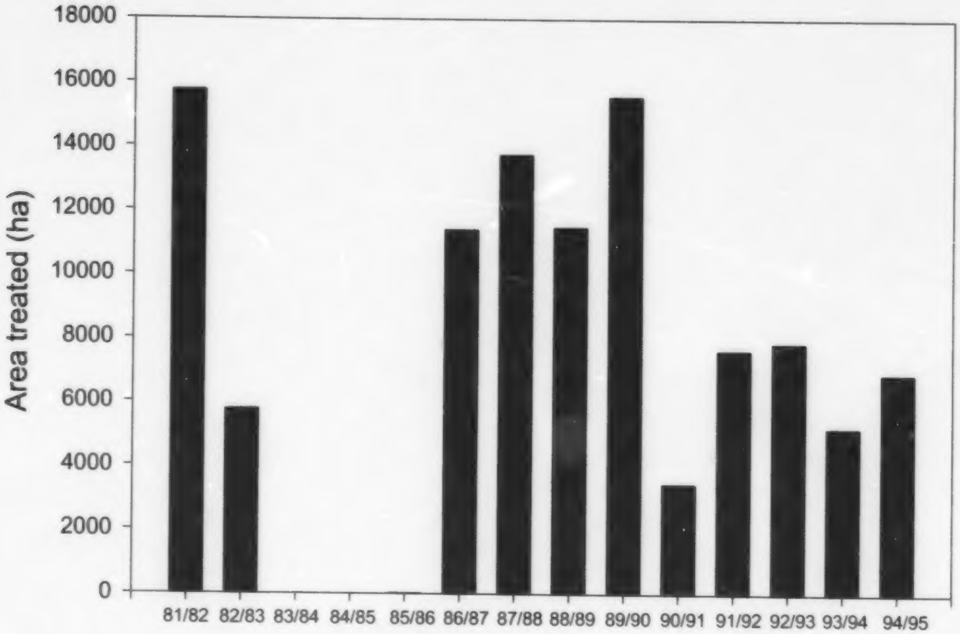


Figure 3.11. The area of Crown land in British Columbia treated with N fertilizer, 1981-1995. (Data source: BCF 1997).

windrowing of logging slash and forest floor organic layers. Actual nutrient losses, through leaching to surface waters after site preparation, have not been quantified.

In British Columbia, the increase in first-order stream nitrate concentrations following slashburning was substantial, relative to the small response to clearcutting (Feller and Kimmins 1984). The ash raised the pH of strongly acid forest soils, thereby favouring increased rates of nitrification in the soil. Leaching or washing of ash into streams, after wildfire, has also been observed. However, in all cases the highest nitrate concentrations observed were well below drinking water standards.

Fertilization

Nitrogen and phosphorus fertilizers are not applied to natural or plantation forests throughout most of Canada, for economic reasons. In British Columbia, however, they have been applied to a limited extent to both coastal and interior conifer forests on Crown (Figure 3.11) and private land, to increase wood production on medium and good sites. Nitrogen is applied at rates of 150 to 250 kg/ha, but rarely more than once during a crop rotation. Based on 1994-95 data of approximately 6800 ha treated with N fertilizer and assuming an application rate of 200 kg/ha N, then a total of 1.4 thousand tonnes N were used to fertilize forests in British Columbia. In a summary of fertilization research from the Pacific northwest USA, Binkley and Brown (1993) concluded that N fertilizers do not lead to excessive nitrate levels in streams. Two exceptions are studies in British Columbia and New Brunswick where N application resulted in short-lived substantial increases in stream nitrate that approached values of concern for human health. In these studies no efforts were made to minimize application over streams, so initial nitrate increases resulted from stream interception of aerial applied N. Even when inorganic N concentrations in streams (Table 3.17) and soils were significantly elevated in the year of application, they were rapidly dissipated thereafter (Jewett et al. 1995). Up to 90% of the N applied was retained by vegetation and soil or volatilized from the soil surface when urea was applied to a boreal pine forest (Morrison and Foster 1977).

In summary, the limited data indicate that nutrient losses from forest management practices have generally been small relative to those in harvested biomass or in the smoke of slash burns. There is, however, insufficient information on nutrient leaching from Canadian forests to generalize about nutrient losses associated with forest management practices. Additional research is required to quantify nutrient losses from terrestrial systems to ground and surface waters following timber harvest for the diversity of forested ecozones in Canada.

3.6 Atmospheric Transport and Deposition

Pollutants in the atmosphere may be transported by winds and global air circulation anywhere from a few tens of metres to thousands of kilometres beyond their point of release before they are deposited. Atmospheric deposition refers to wet or dry deposition of a chemical from the atmosphere to the Earth's surface by physical processes. Wet deposition is the removal of chemicals from the atmosphere by precipitation (rain, snow or fog). Forms of nutrients commonly found in wet deposition are ammonium, nitrate and phosphate as well as organic N and P. Dry deposition refers to any process that does not involve precipitation, such as the settling of particles of dust or soil or the incorporation of gaseous compounds into plants or water. Turbulent processes in the boundary layer, the chemical and physical nature of the deposition species, and the capacity of the surface to capture or absorb gases and particles govern dry deposition. N and P can be bound to or incorporated into particles (such as eroded soil particles or particles released as a result of industrial process) that enter the atmosphere and are then dispersed, transported and later deposited. Wet deposition has typically received more attention than dry fallout as a source of atmospheric N and P, possibly because of the difficulty in obtaining accurate estimates of dry deposition due to the effects of turbulence on small particles.

Since 1977, Environment Canada and most Canadian provinces have monitored precipitation to determine wet deposition of acid-related substances, commonly known as acid rain (AES-EC 1997). Nitrate and sulfate are the main components of acid precipitation (AES-EC 1997). Nitrate in the atmosphere is derived from the oxidation of nitrogen oxides to nitric acid. Nitrogen oxides are produced naturally through bacterial activity, wildfires and lightening. Their abundance is greatly enhanced by industrial combustion processes that oxidize atmospheric N gas (N_2). Ammonia is released on occasion following transport accidents, from refrigerator systems, near sources of anhydrous ammonia production, and from areas of intensive livestock management (Van der Eerden 1982; Teshow and Anderson 1989).

In Canada, atmospheric deposition arriving from long distance transport supplies approximately 2.5 kg/ha/yr N in the form of nitrate and ammonium, based on 1990s data (Figure 3.12). This value is five-fold greater than pre-industrial values of 0.5 kg/ha/yr N. The increase in wet deposition of nitrate and ammonium in recent decades, both in absolute terms as well as in relation to sulfate, is a major concern particularly in eastern Canada (Grennfelt and Hultberg 1986; Berden and Nilsson 1996; AES-EC 1997). Wet deposition of N is considerably higher in eastern than in western Canada. Ten-year averages (calculated using a kriging method to estimate spatial means) are 3.44 kg N/ha/yr east of the Manitoba-Ontario border (C-U. Ro, Environment Canada, personal communication) compared to 0.80 kg N/ha/yr west of the border. The most recent data (1996) show wet deposition for fourteen

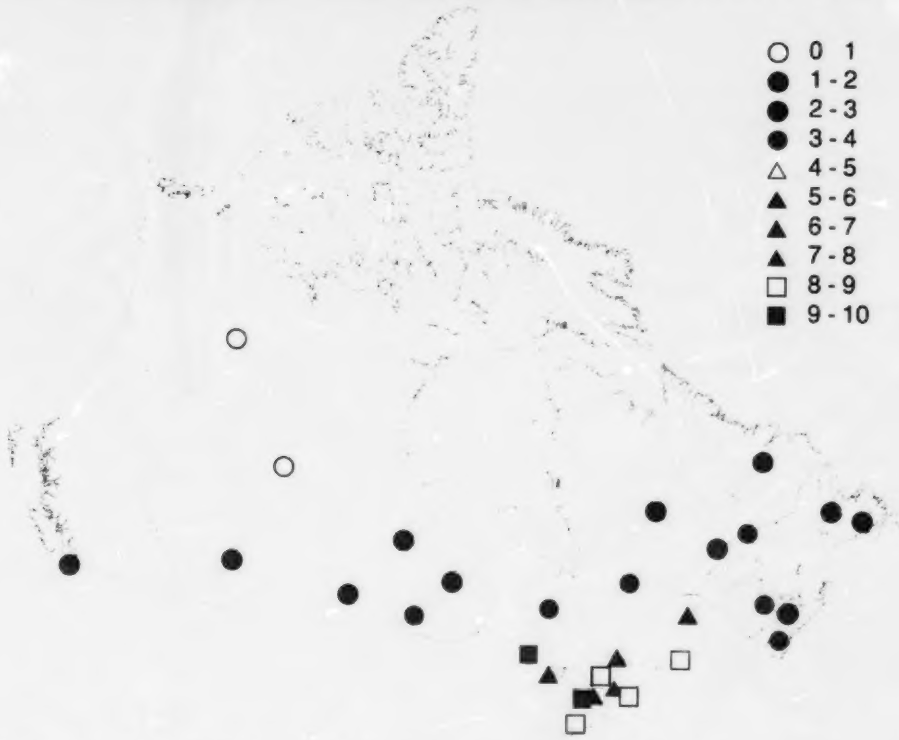


Figure 3.12. Map showing the wet deposition (kg N/ha/yr) of nitrate and ammonium measured at 30 sites across Canada during 1984-1994, based on measurements made by Canadian Air and Precipitation Monitoring Network (Ro et al. 1998).

locations in Canada ranging from 0.25 to 5.09 kg/ha/year for nitrate as N and 0.31 to 3.39 kg/ha/year for ammonium as N (Table 3.18). National data are not available for dry deposition of N; dry deposition, however, may be a considerable source of N. In southern Ontario, total atmospheric N deposition arriving from long distance transport was estimated to be 18.4 kg/ha and included wet deposition of 10.4 kg/ha and dry deposition of 8.0 kg/ha (Barry et al. 1993). Our estimates of atmospheric N deposition include only wet deposition of ammonium and nitrate. Using the 10-year wet deposition rates of 3.44 kg N/ha/yr east of the Manitoba-Ontario border and 0.80 kg N/ha/yr for west of this border, atmospheric deposition of ammonium and nitrate is estimated to add 182 thousand tonnes N to freshwaters (106 223 600 ha), 43 thousand tonnes to cropland (34 918 733 ha), 117 thousand tonnes to agricultural land not used for crops (31 064 291 ha) and 1 378 thousand tonnes to non-agricultural land (835 983 249 ha) in Canada each year.

In contrast to N, phosphorus does not exist in a gaseous form. Phosphorus in the atmosphere is present as particulates (dust and organic debris) and as water-soluble phosphate (PO_4^{3-}). Natural sources of atmospheric P include terrestrial airborne matter, volcanic dust and meteoric particles; anthropogenic sources include releases from fertilizer application as well as industrial sources that produce P products (e.g., fertilizer production, phosphate rock processing, and elemental P production). Atmospheric P loading is not routinely measured. The few Canadian studies to do so have been part of larger projects aimed at developing detailed P budgets for lakes. These studies found that, in Canada, atmospheric P loading due to both wet and dry deposition ranged from 0.01 to 0.74 kg/ha/yr (Table 3.18). Of this deposition, dry fallout of total P accounted for 34-65% in both Alberta and South-Central

Table 3.18. 1996 Annual precipitation, weighted-mean concentration (mg/L), and load (kg/ha/year) of nitrate-N and ammonium-N in wet deposition for various Canadian sites. (Data source: Ro et al. 1998).

CITY	Total Precipitation (cm)	Nitrate-N mg/L	Nitrate-N kg/ha/year	Ammonium-N mg/L	Ammonium-N kg/ha/year
Snare Rapids, NWT	38.92	0.063	0.25	0.078	0.31
Saturna, BC	105.48	0.185	1.95	0.099	1.05
Esther, AB	33.29	0.215	0.71	0.266	0.88
Point du Bois, MB	46.36	0.021	1.20	0.119	2.13
Island Lake, MB	48.44	0.014	0.73	0.046	0.93
Experimental Lakes Area, ON	101.41	0.201	2.04	0.245	2.49
Warsaw Caves, ON	108.74	0.468	5.09	0.312	3.39
Chalk River, ON	93.90	0.404	3.80	0.290	2.72
Mingan, QC	127.22	0.122	1.55	0.059	0.75
Montmorency, QC	163.35	0.221	3.62	0.131	2.14
Harcourt, NB	129.71	0.158	2.05	0.087	1.13
Kejimkujik, NS	162.14	0.151	2.45	0.071	1.15
Bay d'Espoir, NF	154.47	0.077	1.19	0.043	0.66
Goose Bay, NF	112.15	0.066	0.73	0.039	0.44

Ontario (Shaw et al. 1989; Scheider et al. 1979). Typically, atmospheric deposition represents only 1 to 6% of the total P budget in Canadian lakes (Peters 1973; Ahl 1988); however, in the very dilute lakes of northwestern Ontario, precipitation may supply as much as 80% of the P input (Barica and Armstrong 1971).

3.7 Conclusions

This chapter presents information on N and P loading to Canadian water, land and air from household, industrial, agricultural, aquacultural and atmospheric sources. These estimates were obtained using several approaches. In the case of municipal sewage, septic disposal and aquacultural losses to surface waters, manure production by livestock and crop uptake of nutrients, our approach was to multiply census data (population numbers for humans and livestock; production for crops and finfish) by average coefficients for nutrient emission (in the case of humans, livestock and finfish) or nutrient content (in the case of crops). Estimates of industrial and agricultural N releases to the atmosphere (Environment Canada 1997a, Monteverde et al. 1997, Vézina 1997, Desjardins and Keng 1999, Environment Canada 1999a, Desjardins and Reznek 2000) and household compost production (CCC 1998) were calculated using a similar approach by the reporting agency. In the case of industrial loads to water and fertilizer use, measured values are reported. Atmospheric deposition of nitrate and ammonium is based upon measured values from which weight-averaged regional values were calculated by the reporting agency (Ro et al. 1998). All data are based on mid-1990s values, except for estimates of biosolid application to farmland. The most recent national estimates for this are from the early 1980s (OECD 1995). The use of average emission factors and census data to estimate nutrient loading to water, land and air does not allow consideration of the variability around an average value (e.g., due to temporal or regional variation in pollutant loading) and these may be significant. However, at present, pollutant-loading estimates based upon average emission factors and census data represent the best (and most widely used) approach to obtain regional and national estimates of pollutant loading. These estimates allow comparison of the magnitude of N and P loading from various sectors to Canadian water, air and land.

Table 3.19. Annual bulk (wet and dry) deposition (kg/ha/yr) of total phosphorus (TP) for various location in Canada.

Location	TP (kg/ha/yr)	Reference
Arctic (Cornwallis Island)	0.05	Schindler et al. 1974
Saqvaqjuac, NWT	0.01	Welch and Legault 1986
Central Alberta	0.20	Shaw et al. 1989
Northwestern Ontario	0.33	Schindler et al. 1976
	0.25 (wet only)	Linsey et al. 1987
Central Ontario	0.42 (bulk)	Scheider et al. 1979
	0.17 (wet only)	Scheider et al. 1979
	0.39	Jeffries et al. 1978
	0.37	Gomolka 1975
	0.74	Nicholls and Cox 1978
Eastern Ontario	0.30	Schindler and Nighswander 1970

Discharges to Surface and Ground Waters

Household sewage is the largest point source of N and P to the Canadian environment. In 1996, an estimated 5.6 thousand tonnes of total P and 80 thousand tonnes of total N were released to lakes, rivers and coastal waters from MWTPs in Canada (Table 3.20). This load occurred despite the fact that in 1996 73% of Canadians were served by municipal sewer systems and at least 94% of the wastewater collected by sewers received primary or higher treatment. Most of the N and P in household sewage is from human waste (urine and faeces) and is present in forms immediately available for plant uptake. In addition to household sewage collected in sewers, 1.9 thousand tonnes of P and 15 thousand tonnes of N are released from septic sewage systems and make their way to ground water. Stormwater sewers and combined sewer overflows likely also add approximately 2.3 thousand tonnes of P and 11.8 thousand tonnes of N to surface waters; most of these nutrients are present in particulate forms and are therefore not immediately available to plants. There are no national figures for losses due to leaching from municipal landfills.

Discharge of industrial wastewater adds at least 2.0 thousand tonnes of total P and 11.8 thousand tonnes of N (as nitrate and ammonia) to Canadian surface waters (Table 3.20). This number is an under-estimate as data are not available for NB, NS, PEI, and Québec industries that do not discharge to the St. Lawrence River Basin. In addition, not all industries are monitored in all provinces or territories and N is reported not as total N but as ammonia and nitrate. There are also no national figures for losses due to leaching from industrial landfills.

In 1996, approximately 56 thousand tonnes of P and 294 thousand tonnes of N remained in the field after crop harvest (see details in Table 3.20). There is no national information on how much of this residual P and N moves to surface or ground waters. A recent assessment of N losses from agricultural land where the soils have a water surplus predicted that 17% of Ontario, 6% of Québec and 3% of Atlantic farmland would produce runoff or seepage water with > 14 mg N/L. (Macdonald 2000b). In British Columbia, 5% of the agricultural land has a water surplus and 69% of this area was predicted to generate runoff or seepage water with N concentrations > 14 mg/L.

Table 3.20: Comparison of P and N loading to Canadian surface and ground waters from various sources, 1996. Industrial N loads are based on $NO_3 + NH_3$ and not total N; industrial data are not available for NB, NS and PEI and Québec industries that do not discharge to the St Lawrence River. Agricultural residual is the difference between the amount of N or P added to cropland and the amount removed in the harvested crop (see Table 3.22 for detailed calculations); data are not available as to the portion of this residual that moves to surface or ground waters.

Nutrient Source	Total Phosphorus (10^3 t/yr)						
	Atlantic	Québec	Ontario	Prairies	British Columbia	Territories	Canada
Municipality							
MWTPs[1]	0.9	2.1	1.0	0.6	1.0	0.01	5.6
Sewers							2.3
Septic Systems	0.3	0.5	0.6	0.3	0.2	0.01	1.9
Industry	0[2]	0.01	1.0	0.4	0.7	0	2.0
Agriculture (residual in the field after crop harvest)	10	29	18	-19	13	n/a	56
Aquaculture	0.2	0.01	0.04	0.01	0.2	n/a	0.5
Atmospheric Deposition to Water							n/a

Nutrient Source	Total Nitrogen (10^3 t/yr)						
	Atlantic	Québec	Ontario	Prairies	British Columbia	Territories	Canada
Municipality							
MWTPs[1]	4.6	19.9	31.7	13.2	10.6	0.3	80.3
Sewers							11.8
Septic Systems	2.2	3.7	5.0	2.6	1.9	0.05	15.4
Industry	0.1[2]	0.3[3]	9.9	0.6	0.9	0	11.8
Agriculture (residual in the field after crop harvest)	18	46	14	188	29	n/a	294
Aquaculture	0.8	0.04	0.2	0.04	1.2	n/a	2.3
Atmospheric Deposition to Water (NO_3^- N and NH_4^+ N only)	11.9	60.7	54.4	13.9	1.6	39.9	182

[1] MWTPs, municipal wastewater treatment plants

[2] data from Newfoundland only

[3] data for industries discharging to the St. Lawrence River

Aquaculture is a small but growing source of nutrients to Canadian waters. Nutrient release from fish production systems results from the excretion of dissolved or solid waste and from unconsumed feed. Approximately 0.5 thousand tonnes of P and 2.3 thousand tonnes of N are released to coastal and fresh waters from aquaculture operations in Canada (Table 3.20).

Forest management practices that disrupt the cycle of nutrients between the soil and trees (e.g., timber harvest, site preparation and slashburning, and fertilization) may increase stream water concentrations of N and, to a lesser extent, P. However, because the effects have been studied at relatively few sites in Canada, changes in nutrient loading caused by forest management practices cannot be described for most of the country.

Atmospheric Release and Deposition

At least 1 400 thousand tonnes of N is released to the atmosphere each year from anthropogenic sources in Canada (Table 3.21). Of this total, the largest single source of N emissions is agricultural activity, in particular the release of ammonia associated with handling and application of manure and fertilizer. Release of nitrous oxide (N_2O) was split approximately evenly amongst industrial, transportation-related, and agricultural sources. In the case of nitric oxide and nitrogen dioxide (i.e., NO_x), the largest emissions were from industrial and transportation-related sources; NO_x emissions are not available for agricultural emissions but are likely similar to industrial emissions (Janzen et al. 1998). Reliable data are not available for P emissions to air.

Once pollutants are released to the air, they may be transported by winds and global air circulation for great distances before they are deposited on land and water. Nitrate is the nutrient in highest abundance in atmospheric deposition. Nitrogen loads due to wet deposition of nitrate and ammonium average 3.4 kg N/ha/yr east of the Manitoba-Ontario border compared to 0.8 kg N/ha/yr west of the border (Figure 3.12). Nitrate and ammonia deposition is considerably higher in eastern than in western Canada as a result of industrial activities in Central Canada and the northeastern USA. In contrast to N, P deposition is not routinely measured. The few studies conducted found that P inputs from wet and dry deposition together ranged from 0.01 to 0.7 kg/ha/yr.

Table 3.21. Sectoral comparison of N loading from anthropogenic sources to the Canadian atmosphere. Data are 1995 estimates unless indicated otherwise. N_2O data from Environment Canada 1997a except agriculture N_2O data which are from Desjardins and Keng 1999; NO_x data from Environment Canada 1999a; industrial ammonia data from Environment Canada 1999b; other ammonia data from Vézina 1997.

Nutrient Source	**N_2O (10^3 t/yr N)**	**NO_x (10^3 t/yr N)**	**NH_3 (10^3 t/yr N)**
Industrial emissions	27	275	27
Transportation combustion	31	393	4
Agricultural emissions[1]	38[2]	n/a[3]	570
Other	2	82	19
Canadian anthropogenic total	98	750	623

[1] does not include fuel combustion or biomass burning.

[2] 1996 data

[3] possibly similar in magnitude to industrial emissions (Janzen et al. 1998); agricultural emissions not included in total.

Additions to Soils

Fertilizer and manure nutrients applied to agricultural lands are essential to maintain crop yield and soil health. In 1996, 297 thousand tonnes of P and 1576 thousand tonnes of N as fertilizer were applied to cropland in Canada (Table 3.22). In addition, 139 thousand tonnes of P and 384 thousand tonnes of N were applied as manure. Biosolids (the portion of sewage sludge that has been stabilized to meet regulations for land application) may also be applied to agricultural land. Of the estimated 500 thousand tonnes of sewage sludge produced annually in Canada (early 1980s data; OECD 1995), approximately 42% is applied to agricultural land and supplies 8 thousand tonnes of N and 5 thousand tonnes of P. An additional 43 thousand tonnes of N are added to cropland as a result of wet deposition of nitrate and ammonium.

In addition to desirable nutrient inputs to soils (e.g., commercial fertilizer, manure and biosolids), nutrients are also added as a result of waste disposal. An estimated 10.5 million tonnes of residential solid waste were collected by municipalities across Canada in 1992 (Environment Canada 1996c). The majority of residential solid waste is landfilled, with the remainder incinerated, recycled, or composted, in the case of some organic waste. In the case of industrial solid waste, approximately 12 million tonnes were generated in Canada in 1995. Of this, 78% was landfilled, 20% recycled, and the remainder incinerated (Statistics Canada 1998b). Nitrogen and P release from the breakdown of solid waste is not known.

Table 3.22. Sectoral comparison of N and P loading from anthropogenic sources to Canadian soil. Fertilizer, manure and legume data are from 1996; forestry data are from 1994-95; biosolids are data from the early 1980s; compost data are from 1996, and atmospheric data are from 1995. See text for data sources and calculations. Atmospheric deposition includes only NO_3^--N and NH_4^+-N.

Nutrient Source	**TN (10^3 t/yr N)**	**TP (10^3 t/yr P)**
Added to Crop Land		
Fertilizer	1576	297
Manure	384	139
Legumes (total atmospheric N fixed)	773	
Biosolids	8.4	5.3
Atmospheric Deposition (arriving from long distance transport)	43	
Total added	2784	442
Removed with the harvest	2491	386
Difference	293	56
Added to Forested Land	1.4	
Compost	3	5.5
Atmospheric Deposition to Non-Cropland	1495	
Added to Landfill		
Biosolids	3.6	2.3
Solid waste	n/a	n/a

4.0 The Effects of Nutrient Addition on Ecosystems

Highlights

- Addition of nutrients to an aquatic or terrestrial ecosystem increases the biomass of plants and, ultimately, decreases the number of species. Although ecosystems managed for plant yield, such as agriculture and forestry, reap economic benefits from added nutrients, natural ecosystems generally suffer an undesirable change in plant and animal communities.

- Nutrient addition to lakes causes eutrophication, characterized by the development of algal scum (including blooms of toxic algae), changes in the abundance and composition of aquatic animals including an increase in the numbers of "coarse" fish, declines in oxygen in the lake water, and taste and odour problems of drinking water supplies. Examples of eutrophic lake waters are found in some of the sheltered bays of Lakes Erie and Ontario.

- Nutrient addition to rivers increases the abundance of aquatic weeds or algae. This increased plant growth may cause problems to boaters and clog the screens of intake pipes. Many rivers in Canada show signs of moderate nutrient enrichment downstream of municipal wastewater discharges or areas of intensive agriculture.

- Water quality on both coasts is generally excellent, as the ocean's natural circulation serves to stop accumulation of nutrients from anthropogenic sources. Certain small bays and harbours where oceanic exchange is limited do, however, show evidence of eutrophication, especially on the East Coast where settlement has affected the environment for over 200 years.

- Wetlands are unique ecosystems that straddle the boundary between terrestrial and aquatic habitats. Although the conversion of wetlands to urban and agricultural land has been the major cause of wetland loss, a less obvious cause of undesirable changes in wetland plant communities is nitrogen addition. Cootes Paradise is a freshwater wetland in southern Ontario showing signs of recovery now that nutrient addition from municipal sewage has been reduced.

- Nitrate contamination of ground water is usually due to over-application of fertilizer or manure on agricultural land, domestic waste from septic disposal systems and losses of soil organic nitrogen. Contaminated ground water may contribute to the enrichment of some surface waters in Canada.

Nutrient addition, whether it be the purposeful addition of fertilizers to agricultural fields or the unintentional consequence of waste discharge, promotes plant growth in all ecosystems except those few that are nutrient sufficient. Terrestrial plant production is limited by N supply for much of the Earth's surface, particularly in temperate and boreal regions (Vitousek and Howarth 1991). Agricultural crops are a clear example of production gains to be realized from the addition of N, the dominant ingredient in commercial fertilizers. However, native vegetation is also N limited (i.e., maximum growth is prevented by insufficient N). Consequently, native vegetation responds to N addition by increased biomass if not limited by other factors such as moisture, light, or grazing (Rasmussen et al. 1993; Hudson et al. 1994; Aber et al. 1995; Townsend et al. 1996).

In aquatic ecosystems, the addition of inorganic nitrogen to N-limited systems will also increase plant production to excessive levels, a condition known as eutrophication. Most marine waters, particularly those in temperate regions, are N-limited and N addition as a result of human activity is arguably the greatest threat to the integrity of coastal ecosystems (NRC 1993a; Howarth et al. 1996). In contrast, most lakes tend to be limited by P supply (Dillon and Rigler 1974; Schindler 1974). In the case of river and freshwater wetlands, less information is available about their nutrient status; however, evidence to date suggests these systems may be N or P-limited when little affected by human activity (Morris 1991, Scrimgeour and Chambers 2000). As well as increased plant growth resulting as a direct consequence of nutrient addition to inland and coastal waters, indirect consequences include changes in the abundance and diversity of higher trophic levels (benthic and planktonic fauna), increased abundance of toxic algae, and loss of oxygen from the water. This last effect is due to increased oxygen consumption by the greater abundance of bacteria supported by decomposition of nutrient-enhanced plant production.

The primary consequence of nutrient addition is similar for all ecosystems, namely increased plant growth. However the secondary consequences (e.g., changes in faunal diversity, effects on nutrient cycling, impacts on adjacent ecosystems) differ. This chapter reviews the effects of nutrient addition on forests, lakes, rivers, wetlands, coastal waters and ground waters. Canadian case studies are presented to support the general reviews, and the geographic extent of adverse consequences to nutrient addition is identified.

4.1. Terrestrial Ecosystems

Nitrogen is considered to be the nutrient that most often limits net primary production in forest ecosystems, particularly those in temperate and boreal regions (e.g., Vitousek and Howarth 1991). Fertilization studies involving single N applications have confirmed that N is the major limiting element for tree growth in many Canadian forest ecosystems (Weetman et al. 1987; Morrison and Foster 1990). Not all forests in Canada are N limited, however. Maple-dominated deciduous forests in eastern Canada have not responded to added N, likely because of an abundance of soil N (Cote et al. 1993; I.K. Morrison, Natural Resources Canada, personal communication). However, the general tendency of N limitation in terrestrial ecosystems, together with the documented increase in atmospheric N deposition as a result of human activities, have raised the question as to how much of the carbon stored in terrestrial ecosystems is a consequence of anthropogenic N deposition and fixation.

The primary source of N to forest ecosystems is the atmosphere. Prior to industrial development, N forms taken up by plants were produced only by biological nitrogen fixation (i.e., microbially mediated production of ammonia) and lightning, which results in the production of nitric oxide, NO. Although plants have the capacity to remove nitrogen oxides from the atmosphere (Bennet 1975; WHO 1997a), most N is taken up from the soil where it has been fixed in biologically available forms by N-fixing bacteria.

Human activity has greatly increased emissions of airborne N that, in turn, have led to increased deposition of biologically-available N on the Earth's surface. Experiments conducted in Europe and North America indicate a large portion of this extra anthropogenic N is retained by forest, wetland, and tundra ecosystems and stimulates production, resulting in increased carbon uptake and storage (see Vitousek et al. 1997 for discussion). N deposition can also stimulate microbial decomposition and thus

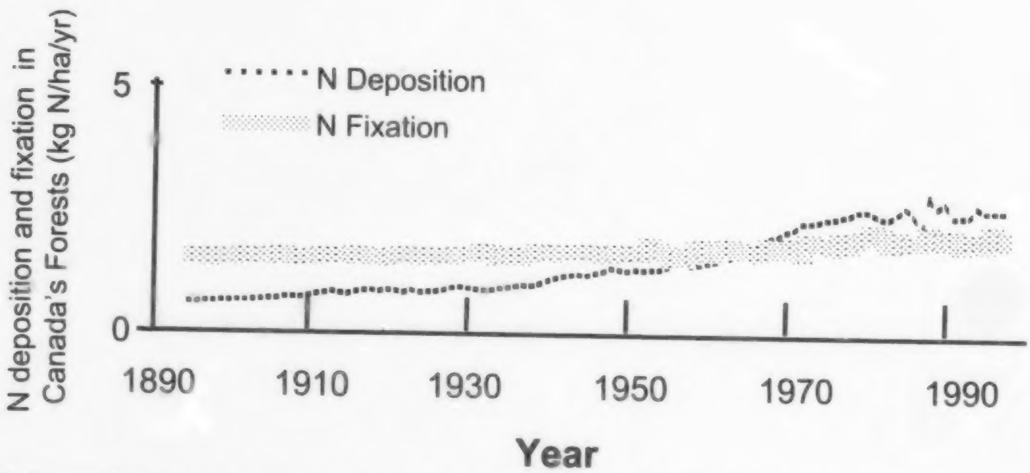


Figure 4.1. Atmospheric N deposition in Canada and N fixation by Canada's forest ecosystems (from Chen et al. 2000 a, b).

the breakdown of any N-enhanced productivity. However, the stimulatory effects of N appear, on average, to out-weigh productivity losses on a global basis. Global estimates of additional plant growth (expressed as carbon storage) resulting from anthropogenic N addition range from 1×10^8 to 13×10^8 tonnes of carbon, with higher numbers tending to reflect our better understanding in recent years of the impact of humans on the global N cycle (Peterson and Melillo 1985; Schindler and Bayley 1993; Hudson et al. 1994; Townsend et al. 1996).

In Canada, atmospheric deposition of inorganic N (ammonium plus nitrate) has steadily increased since the 1900s until the 1990s when it was estimated to supply, on average, 2.5 kg/ha/yr of N to forests (Figures 3.14 and 4.1). This value is five-fold greater than pre-industrial values of 0.5 kg/ha/yr N. Because most of Canada's forests are N limited, increased N deposition has likely increased the productivity of forest vegetation. It is still not clear, however, if productivity gains in Canada's forests resulting from increased N deposition, along with a warming climate and elevated CO_2 levels in the atmosphere, have offset the carbon loss caused by timber harvest, fire and insect-induced mortality. Kurtz and Apps (1999) estimated Canadian forest ecosystems were a sink for atmospheric carbon from 1920 to 1980 but became a carbon source in the 1980s as a result of stand-replacing disturbances, largely of natural origin, that reduced the quantity of stored carbon (Figure 4.2b). In contrast, Chen et al. (2000a, b) estimated that the productivity gains resulting from increased N deposition (about 70 million tonnes carbon per year from 1985 to 1998), along with a warming climate and elevated CO_2 levels in the atmosphere, have more than offset the carbon loss caused by timber harvest, fire and insect-induced mortality, so that Canada's forests are now a carbon sink (Figure 4.2a).

In addition to increases in forest growth, N deposition can also change species dominance and diversity by favouring the growth of plants better adapted to high concentrations of N. Experiments involving N addition have resulted in declines in the number of plant species in North American grasslands, European grasslands and European heathlands (e.g., Aerts and Berendse 1988; Bobbink et al. 1988; Tilman 1997). Moreover, the resulting species-poor plant communities experienced greater variations in productivity in response to environmental stress such as interannual climatic variation (Tilman and

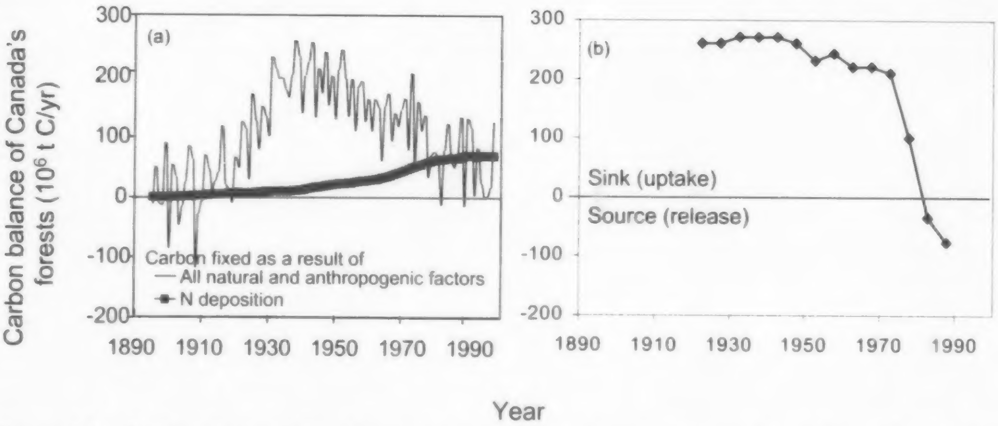


Figure 4.2. Carbon balance for Canada's forest ecosystems: (a) results of Chen et al. (2000a); (b) results of Kurz and Apps (1999).

Downing 1994; Tilman 1996). In Canada, there is little if any conclusive evidence of changes in forest diversity as a result of N addition (Hall 1995).

Although N is the element that most often limits net primary production in terrestrial ecosystems, a point may ultimately be reached where N supply as a result of N deposition is in excess of the total plant and microbial demand for N. This condition is known as forest ecosystem N saturation and is characterized by the conversion of excess N to nitrate, which is then leached to surface or ground waters. In addition to its effects on downstream waters, nitrate leaching has been found to cause changes in soil chemistry, in particular a loss of nutrient cations (such as calcium and magnesium), an increase in soil acidity, and an increase in the availability of aluminum (due to the increased acidity) (see Aber et al. 1998 for review). Leaching of nitrate and associated cations can, in turn, cause nutrient imbalances in trees, in particular changes in calcium:aluminum and magnesium:nitrogen ratios. These imbalances have been linked to reductions in net photosynthesis, photosynthetic N-use efficiency, forest growth, and tree mortality (Schulze 1989; McNulty et al. 1991, 1996; Aber et al. 1995; Cronan and Grigal 1995). Nitrogen imbalances in trees as a result of high levels of N deposition may also increase the likelihood of damage caused by weather extremes, particularly high temperatures in late winter, hard frosts in early spring or prolonged drought (Cote and Ouimet 1996).

Much of the research on forest ecosystem N saturation has been undertaken in Europe and several studies have concluded that the forests of Europe are becoming N saturated. The risk of N saturation in North America is generally lower because most forest ecosystems are N limited and because N deposition is still relatively low (averaging about 2.5 kg/ha/yr in Canada; Figure 4.1). By comparison, forests in some parts of northern Europe are receiving 10 times the natural level of atmospheric N deposition, with the Netherlands receiving the highest rates of atmospheric N deposition in the world (40 to 90 kg/ha/yr; Vitousek et al. 1997). Nevertheless, studies in the USA have shown that certain forests are already N saturated (e.g., Fernow experimental forest, West Virginia; Gilliam et al. 1996) and that other forests could be experimentally N saturated by several years of low-level additions of N (e.g., in a high-elevation spruce-fir stand, Vermont [McNultry et al. 1996]; Bear Brook, Maine [Kahl et al. 1993]; Harvard Forest, Massachusetts [Aber et al. 1993]).

In Canada, an analysis of nitrate concentrations in runoff water to Turkey Lake in central Ontario, Lac Laflamme in southern Québec, Plastic Lake in south-central Ontario, the Experimental Lakes Area in northwestern Ontario, and Kejimkujik Lake in southern Nova Scotia identified the forests surrounding Turkey Lake and Lac Laflamme as being in an early stage of N saturation (Jeffries 1995). The lack of retention of atmospheric N by vegetation (maple-dominated forest) and soils in the Turkey lakes watershed is due to the near-N saturated soils that exhibit high rates of endogenous nitrification (Mitchell et al. 1992). Depletion of base cations from forest soils has also been observed in areas of high sulphate and nitrate deposition (Houle et al. 1997; Markewitz et al. 1998) and predictions based on cumulative loss of base cations indicate forest productivity will decline with continued sulphate and nitrate deposition (Arp and Oja 1992; Arp et al. 1996). The risk of N saturation is greatest in eastern Canada where N deposition rates are highest, with wet deposition being about 10 kg/ha/yr and total deposition (wet + dry deposition) being 13.5 kg/ha/yr (Chen et al. 2000a).

4.2. Lakes

Nutrients are essential to lakes because they provide the raw material for the growth of algae the food sources of zooplankton, which, in turn, are eaten by fish. The concentration of nutrients in a lake is determined by the interplay of the magnitude, timing and bioavailability of the nutrient load, the rate of water supply compared to the volume of the lake (flushing time), and the depth of the lake. Lakes have a capacity to absorb nutrients but when this level is exceeded concentrations rise and obnoxious growths of algae and other effects occur.

Phosphorus in lakes is highly reactive, binding with soil particles and minerals in the water column. Once this binding has occurred the P is largely unavailable to bacteria and algae for growth. It is the amount and rate of turnover of the remaining dissolved phosphorus that dictates the extent of algal growth. The ability of a lake to retain P depends on net sedimentation, which in turn depends on the flushing rate in the lake as well as the depth. Rapidly flushed lakes reflect the concentration of the inflowing waters. In contrast, the P concentration of a lake that flushes very slowly is largely determined by its sedimentation characteristics (Vollenweider 1976). However, even in lakes with high nutrient retention and low average concentrations, there can be nutrient problems. These problems occur because nutrients are loaded in the nearshore via sewage outfalls and rivers where dilution due to the physical movements of water in lakes is minimal. Although dilution is sometimes viewed as an undesirable solution to pollution, dilution is a key part of the nutrient assimilation capacity of lakes where effluents may have 10 to 100 times the P concentration desired in the ambient water. The concentration of nutrients is the crucial factor in nutrient pollution effects and dilution is critical to amelioration of pollution effects.

Nutrient supply strongly affects the growth of aquatic plants in lakes, particularly algae. In the majority of North temperate lakes, P is the nutrient most in demand, and algal growth and biomass is P-limited (Vollenweider 1976). Thus, in many lakes, algal growth is a function of P concentration (e.g., Watson et al. 1997). However, in some lakes, primary production is dominated by macrophytes, which tend to suppress algae, resulting in clear water even at high nutrient loads (Beklioglu and Moss 1996; Weisner et al. 1997). Lakes can be classified based on their nutrient or "trophic" status. Oligotrophic lakes have very low nutrient concentrations (typically less than 5 μg/L average total P). In these systems, algae are nutrient-limited and low in abundance; hence, the water is typically clear. Oligotrophic water can, however be highly coloured by humic matter. Moderately productive lakes (about 10-30 μg/L total P)

Lake Erie, Ontario: Recovery of a "Dead" Lake

Lake Erie is one of the most widely recognized examples of restoration of an aquatic ecosystem damaged by excessive nutrient loading. The damage began in the 1800s when land clearing for agriculture and settlement increased soil erosion and thus P loading to the lake. For one hundred years, development continued with accelerations occurring in the periods of the World Wars. Dramatic changes began in the 1940s as more and more people were connected to sewage systems that discharged to the lake and detergents with high P content came into use.

Concern about the persistent foam from detergents, the increasing degradation of visible water quality, and increasing general concern for the environment led to scientific studies of Lake Erie by federal government agencies. Surprisingly, there had been few previous studies. Investigations by U.S. agencies and Canadian universities in the late 1920s, early 1950s and early 1960s raised concerns about water pollution and disappearance of fish. In 1970, a bi-national study of the lake confirmed that there had been an increase in nutrient concentrations in the lake and that nuisance algae were appearing. Filamentous algae (*Cladophora*) washed up on shores in great quantities. Excessive algal populations sunk to the bottom where decomposition occurred such that the bottom waters of the central basin were depleted of oxygen by late summer; this caused regeneration of P from the sediments. There was a fear that the P regeneration would lead to an autofertilization condition (Burns 1976).

Results of modelling exercises to predict a safe P load were startling. Annual P loads needed to be reduced from a high of about 28 000 tonnes to about 11 000 tonnes to achieve reasonable algal abundance and restore year-round aerobic conditions in the hypolimnion of the Central Basin. Governments were deeply concerned by the evidence that the huge Great Lakes system could be so affected by anthropogenic nutrient loads. Eventually, the Canada U.S. Great Lakes Water Quality Agreement (GLWQA) was signed in 1972. The agreement incorporated the scientific advice to decrease the P load to 11 000 tonnes per year.

Four main strategies were employed to reduce the P load: (1) P in household laundry detergents (which accounted for about 25% of the P in sewage) was rapidly phased out; (2) sewage treatment plants (MWTPs) were constructed where none existed and primary treatment plants were upgraded to secondary treatment; (3) P precipitation by addition of alum, ferric chloride or other agents was employed to enable all MWTPs handling more than 3.8 million litres per day (one million U.S. gallons) to decrease effluent P to 1 mg P/L or less; (4) non-point source P from urban runoff and agriculture was addressed with ongoing changes in farm practices that reduce runoff and erosion. Although an accurate estimate of the expenditures is not possible, it is accepted that on the order of $15 billion U.S. was spent on the various infrastructure construction and improvement projects.

The results provide a dramatic example of how concerted action can reverse environmental damage. The load of total P to Lake Erie was reduced by more than 50% by the mid-1980s and now oscillates around the recommended 11 000 tonnes annually (Figure 4.3). Much of the damaging P load came from cities discharging sewage into the West Basin of the lake, which is where the main repair was effected as the P concentration in the water decreased from 40 µg P/L to about 20 µg P/L. The P response in the Central Basin offshore has been more modest because the basin is larger and nutrient loads were relatively smaller as the majority of controllable municipal sewage discharges are located in the West Basin.

About the time the P reduction goals were achieved, the zebra mussel (*Driessena polymorpha*) invaded the lake. Prodigious filter feeders, the mussels remove enormous quantities of plankton from the water column and deposit their waste to the sediments resulting in P reductions in the water and improved water quality. Because the mussels can have effects on water quality similar to nutrient reduction and lakes take some time to equilibrate, it is important to distinguish the effects of the load reductions from those of the mussels. Comparison of water quality over the period of nutrient controls showed that P concentrations were about 20% lower in the East and Central Basins in 1988 than in 1968 (Table 4.1). This decrease corresponded to about a 50% reduction in phytoplankton abundance (measured as chlorophyll *a* concentration) and a 50% improvement in water clarity (measured as Secchi disc transparency). Nitrate increased by more than 300% over the same period due to fewer algae to assimilate N and continued discharge of sewage N and agricultural fertilizers. In comparison to the period of nutrient controls, changes following the introduction of zebra mussels have been smaller except for P (Table 4.1). Although P declined to a record low concentration in 1995, there has been a tendency in the last few years (1996-1999) for a recovery back to pre-mussel concentrations. Data are less available for the West Basin and there was much variability due to resuspension of sediments; however, chlorophyll a declined by 76% by 1988 and declined by a further 34% by 1994/95/96. Thus, we do not yet know what the long-term impact of the mussels will be or even if the impact will be stable (Charlton et al. 1999). The mussels have caused a marked increase in

Lake Erie concluded

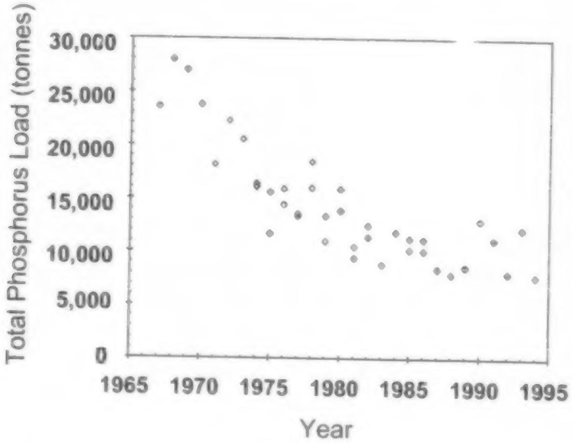


Figure 4.3. Phosphorus loading history of Lake Erie, 1967 to 1994. (Data sources: Fraser 1987; Lesht et al. 1991; D.M. Dolan, International Joint Commission, personal communication).

water clarity in nearshore areas especially in the west basin of the lake (116%).

Despite the success of nutrient controls in lowering phytoplankton abundance, the problem with filamentous algae, (*Cladophora*) has been reduced but not eliminated. Treated sewage contains P at 100 times the desired concentration of the lake water. In mixing zones near sewage outfalls and in nearshore areas receiving flows from rivers, P concentrations are sufficiently high to cause nuisance accumulations of *Cladophora*. Indeed, P concentrations in most Ontario tributaries exceed the provincial guideline of 0.030 mg P/L. Thus, even though the offshore water is of good quality, the conditions in specific locations can still be improved. Now with the advent of the mussel invasion there is some evidence that the excretion of the mussels helps supply nutrients to the attached *Cladophora* algae.

Table 4.1. Percentage change in phosphorus (P), nitrate (N), chlorophyll *a* (Chla), and Secchi depth (Secchi) in the East and Central Basins of Lake Erie during the period of phosphorus reductions (1968 to 1988) and after the zebra mussel invasion (1988 to 1994/95/96).

Basin	1968 to 1988				1988 to 1994/95/96			
	P	N	Chla	Secchi	P	N	Chla	Secchi
East	-19	+325	-43	+48	-33	+53	-25*	+8*
Central	-23	+340	-54	+59	-23	+5*	-17	-5*

* Change is not statistically significant

The success of the nutrient controls in ameliorating the oxygen depletion of the Central Basin hypolimnion has been more equivocal. Beginning in 1993 and until 1997, oxygen concentrations in the hypolimnion were much higher than usual, raising hopes of a recovery in deep-water oxygen (Charlton 1994). However, in 1998, oxygen concentrations throughout most of the Central Basin were less than 1 mg/L oxygen by the end of August, typical of much earlier years. The situation continues to be a subject of research and monitoring.

Despite the inertia of the oxygen situation, the GLWQA nutrient controls have been a success. It is likely that the lake never fully equilibrated with the brief period of peak nutrient loadings so that damage was averted as well as reversed by the substantial reduction in P loads. Further reduction in loads will come with better land use practices; non-point sources now account for about half the total P load. Dealing with human sewage is a perpetual management problem. Maintenance of the hard won gains will be important to preserve the lake. Phosphorus removal at sewage plants increases operating expenses and populations continue to grow. Thus, it is important to maintain monitoring of loads and the performance of sewage plants.

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are said to be mesotrophic (see Heney Lake case study), whereas eutrophic lakes are very rich in nutrients, such that the water is green with algae throughout most of the growing season (see Qu'Appelle Lakes case study). At very high nutrient concentrations (generally about 100 µg/L total P), algal growth is independent of P concentration and becomes limited by other factors such as N, micronutrients (e.g., iron) or light, due to self-shading by high algal densities. Lakes with extremely high nutrient concentrations and excessive algal growth (such that the water can resemble pea soup) are termed hyper-eutrophic.

There are many deleterious effects of algal blooms. Initially, the most noticeable effects are increased water turbidity, reduced aesthetic appeal and even decreased recreational use, as visibility becomes insufficient for swimming. As enrichment continues, erratic and sometimes severe outbreaks of different algal taxa (e.g. Chrysophyta, Haptophyta, Dinophyta) can not only negatively affect water quality and the aquatic food web, but can cause fish kills (e.g., Adachi 1965; Kamiya et al 1979; Hansen et al. 1994; Hallegraeff et al. 1995; Leeper and Porter 1995), particularly, it seems, in acidified systems (Mills et al. 1995). Highly eutrophic systems tend to be predominated by blue-green algae that form dense, foul-smelling and noxious blooms, often as surface scums, because many of these algae are buoyant. These algae can even develop massive blooms under ice-cover, as they are highly efficient at capturing light. Many species of blue-green algae produce potent toxins, which can poison zooplankton, fish, avian waterfowl, terrestrial wildlife, livestock and even humans (Carmichael 1992; Kotak et al 1993a, 1994). Paradoxically, some blue-green algae may at times give the impression of greater water clarity, because their cells are amassed into macroscopic colonies, allowing more light penetration than a similar biomass of individual algal cells (Vollenweider 1976). More frequently, however, surface blooms of blue-green algae can shade out other algae or aquatic plants. Blue-green algae may have other negative effects on aquatic food webs, because they are too large for ingestion by most grazers. This effectively provides them with an escape from predation, and facilitates their growth and dominance, as grazers selectively remove smaller and less noxious algal species (Watson and McCauley 1988).

High algal densities increase costs for drinking-water filtration, and are associated with taste and odour problems. The decline and senescence of algal blooms can lower oxygen concentrations and generate toxic levels of ammonium, thereby shrinking the strata of water habitable by fish. On shorelines with sufficient rocky substrates and P concentrations, attached filamentous algae such as *Cladophora* can produce dense, long filamentous strands that crowd out other species, and provide surface area and nutrients for the growth of odourous mats of other algae and bacteria. These filaments break off during late summer storms and are concentrated on shores by wind action, sometimes to a thickness of over one metre, and extending a substantial distance from shore. This action not only precludes swimming and other forms of recreation, but subsequent decay generates odour and noxious ammonia levels, which may render adjacent property unusable and without value.

Algae (as well as bacteria such as actinomycetes) can cause widespread taste and odour in drinking water supplies. For example, many blue-green algae and actinomycetes produce the compounds geosmin (trans,trans-1,10-dimethyl-9-decalol) and MIB (2-methylisoborneol) which cause musty, earthy odours and tastes in drinking water (Jensen et al. 1994; Jüttner 1995). These two potent volatile organic compounds (VOCs) do not respond to conventional water treatment and their removal costs the water industry millions of dollars per year. Traditionally these VOCs have been associated with intense growths of nuisance algae in highly eutrophic lakes (Brownlee et al.1984; Kenefick et al. 1992; Watson

Qu'Appelle Lakes, Saskatchewan: Effects of Human Activity on Naturally Eutrophic Lakes

Lakes in the Prairies and Boreal Plains ecozones of southern Alberta, Saskatchewan and Manitoba lie in large, fertile agricultural catchments that supply high concentrations of nutrients. Eutrophication is generally the single most important water quality issue (Government of Canada 1996; Hall et al. 1999). Water quality in prairie lakes is controlled by the interplay of climate, resource use, and urban factors. The lakes of the Qu'Appelle Valley in south-central Saskatchewan are a good example of the combined effects of agricultural activity, human settlement, and climate on water quality.

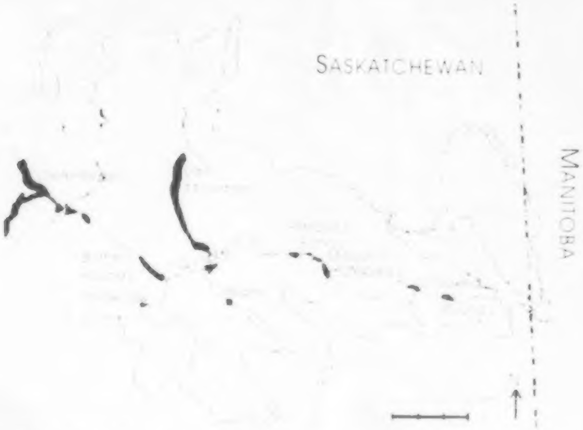

Figure 4.4. Qu'Appelle Valley drainage basin and lakes. Adapted from Hall et al. 1999.

The Qu'Appelle Valley extends over 400 km from its headwaters near Lake Diefenbaker in western Saskatchewan to its confluence with the Assiniboine River in western Manitoba (Figure 4.4). Its 52 000 km^2 basin is drained by the Qu'Appelle River and it, or its tributaries, provides water to approximately one-third of Saskatchewan's population, including the cities of Regina and Moose Jaw. Agricultural fields and pastures comprise more than 95% of land use in the drainage basin (Hall et al. 1999). A chain of eight lakes, including two headwater reservoirs (Lake Diefenbaker and Buffalo Pound Lake), a central group of four 'Fishing Lakes' (Pasqua, Echo, Mission and Katepwa lakes), and two eastern basins (Crooked and Round lakes) form a trophic status gradient in the valley with nutrient concentrations in the lakes increasing from the western to the eastern end of the chain. A ninth lake, Last Mountain Lake, drains into the Qu'Appelle River via Last Mountain Creek. These lakes represent a major recreational and economically-valuable resource for southern Saskatchewan as they are used for commercial (whitefish, cisco and buffalofish) and game (northern pike, walleye, yellow perch) fishing, recreation, irrigation, livestock watering, drinking water supply and sewage discharge, in addition to serving as flood control and waterfowl habitat (Munroe 1986, Chambers 1989).

The lakes of the Qu'Appelle Valley are glacial outwash features (Hall et al. 1999). Typical of this region, the lakes are shallow (mean depth 3.0-14.3 m), hypereutrophic (TP>300 μg/L) and produce immense blooms of blue-green algae throughout the summer (Munroe 1986; Kenney 1990; Hall et al. 1999). Their water quality is modestly influenced by annual fluctuations in precipitation and evaporation rates during the hot, dry summer months. Depending on the amount of rainfall in a given year, their salinity fluctuates between slightly saline (0.5 - 1.0 g/L total dissolved solids (TDS)) and moderately saline (1.0 - 5.0 g/L TDS). Lake Diefenbaker and Buffalo Pound Lake contain the freshest water (TDS < 0.5 g/L) whereas Last Mountain Lake is the most saline with TDS concentrations regularly exceeding 1.5 g/L (Chambers 1989; Hall et al. 1999).

Although the lakes are naturally eutrophic, present water quality is worse than before permanent settlement and intensive agricultural development of the region (Allan et al. 1980; Hall et al. 1999). Growing concern over the deterioration of water quality in the Qu'Appelle lakes during the last 30 years resulted in several Federal-Provincial studies that attributed excessive algal and plant growth to high nutrient concentrations in agricultural runoff and municipal wastewater discharge. At that time (1972), it was estimated that 70% of the P and N entering the river basin came from sewage discharge from

Qu'Appelle Lakes concluded

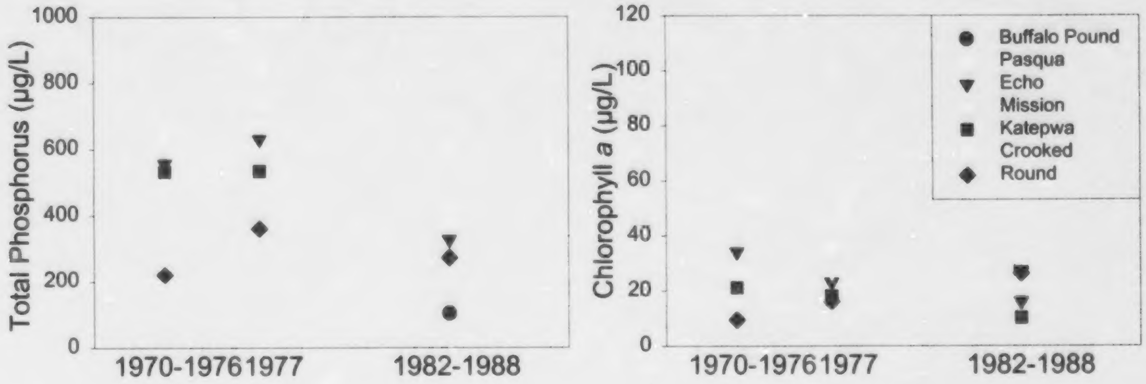

Figure 4.5. Total phosphorus and phytoplankton chlorophyll *a* concentrations for the Qu'Appelle Lakes before (1970-76) and after (1977; 1982-1989) upgrades at the Regina wastewater treatment plant. (Data from Chambers 1989).

Regina and Moose Jaw's wastewater treatment plants (Munroe 1986). As a result, the 1975 Qu'Appelle Agreement between the Governments of Canada and Saskatchewan set out to protect and improve the Qu'Appelle Valley's environment, resources and cultural heritage, and to promote economic growth in the region by developing tourism and recreational activities (Chambers 1989). Regina upgraded its sewage treatment facility to tertiary treatment in 1976 and Moose Jaw diverted all of its sewage to agricultural land through the use of spray irrigation by 1987 (Chambers 1989).

Tertiary wastewater treatment has not resulted in a clear improvement in water quality in the Qu'Appelle lakes. Comparing data collected before and after upgrades to Regina's wastewater treatment plant, open-water total P concentrations have decreased; however, chlorophyll *a* levels have not changed (Figure 4.5; Chambers 1989). Hall et al. (1999) also noted no clear improvement in water quality following sewage treatment despite a three-fold decrease in P loading. However, paleolimnological studies of environmental factors related to long term changes in water quality in the Qu'Appelle Lakes have identified that algal community composition is more variable in the years following tertiary treatment of Regina's sewage (ca. 1977-1990) as compared to a preceding era of high algal production (ca. 1925-1960; Dixit et al. 2000).

N loading to the Qu'Appelle lakes is currently at an all-time maximum (Hall et al. 1999). Nutrient enrichment bioassays (Graham 1997), analysis of elemental ratios (TN:TP; Munroe 1986) and the presence of N_2-fixing blue-green algae (Graham 1997) suggest primary production in the Qu'Appelle Valley is controlled by the influx of N, not simply P. Hall et al. (1999) concluded future management strategies must take into account that water quality in the upstream lakes is influenced by urban activities whereas downstream lakes are more affected by resource use (cropland area and livestock biomass).

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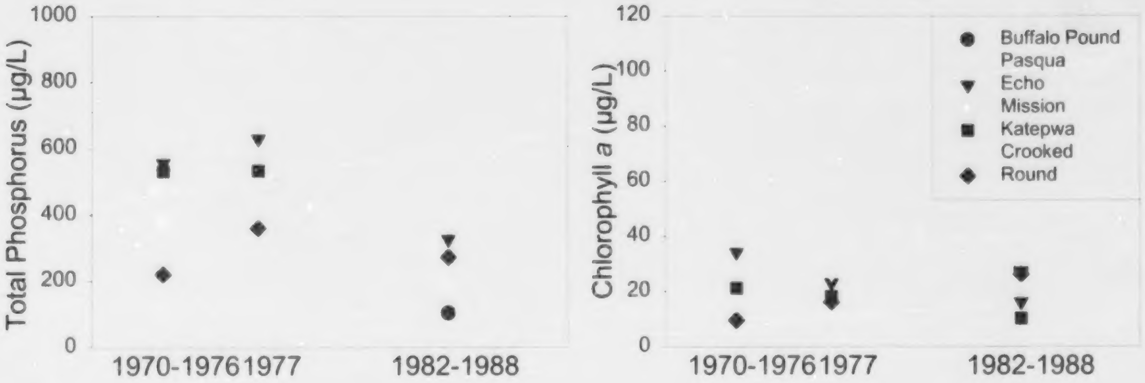
et al. 1999), but more recently, they have been problematic in systems that have undergone nutrient abatement. High levels of these compounds were detected in water from Lake Erie, the Niagara River, eastern Lake Ontario and the upper St. Lawrence River (Lange and Wittmeyer 1997; Ridal et al. 1999; Watson et al. 2000). Odour outbreaks have increased in the Great Lakes and St. Lawrence River system coincident with the invasion of zebra mussels and it has been postulated that the greater frequency of these problems is due to increased benthic algal abundance caused by greater water

Qu'Appelle Lakes concluded



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Regina and Moose Jaw's wastewater treatment plants (Munroe 1986). As a result, the 1975 Qu'Appelle Agreement between the Governments of Canada and Saskatchewan set out to protect and improve the Qu'Appelle Valley's environment, resources and cultural heritage, and to promote economic growth in the region by developing tourism and recreational activities (Chambers 1989). Regina upgraded its sewage treatment facility to tertiary treatment in 1976 and Moose Jaw diverted all of its sewage to agricultural land through the use of spray irrigation by 1987 (Chambers 1989).

Tertiary wastewater treatment has not resulted in a clear improvement in water quality in the Qu'Appelle lakes. Comparing data collected before and after upgrades to Regina's wastewater treatment plant, open-water total P concentrations have decreased; however, chlorophyll *a* levels have not changed (Figure 4.5; Chambers 1989). Hall et al. (1999) also noted no clear improvement in water quality following sewage treatment despite a three-fold decrease in P loading. However, paleolimnological studies of environmental factors related to long term changes in water quality in the Qu'Appelle Lakes have identified that algal community composition is more variable in the years following tertiary treatment of Regina's sewage (ca. 1977-1990) as compared to a preceding era of high algal production (ca. 1925-1960; Dixit et al. 2000).

N loading to the Qu'Appelle lakes is currently at an all-time maximum (Hall et al. 1999). Nutrient enrichment bioassays (Graham 1997), analysis of elemental ratios (TN:TP; Munroe 1986) and the presence of N_2-fixing blue-green algae (Graham 1997) suggest primary production in the Qu'Appelle Valley is controlled by the influx of N, not simply P. Hall et al. (1999) concluded future management strategies must take into account that water quality in the upstream lakes is influenced by urban activities whereas downstream lakes are more affected by resource use (cropland area and livestock biomass).

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et al. 1999), but more recently, they have been problematic in systems that have undergone nutrient abatement. High levels of these compounds were detected in water from Lake Erie, the Niagara River, eastern Lake Ontario and the upper St. Lawrence River (Lange and Wittmeyer 1997; Ridal et al. 1999; Watson et al. 2000). Odour outbreaks have increased in the Great Lakes and St. Lawrence River system coincident with the invasion of zebra mussels and it has been postulated that the greater frequency of these problems is due to increased benthic algal abundance caused by greater water

Lac Heney, Québec: Impacts of Aquaculture (1993-1998)

In November 1998, the Québec government closed a fish farm that had opened in 1993 on a tributary of Lac Heney, Québec (Bird and Mesnage 1996). In five years of operation, the fish farm's effluent was implicated in the near doubling of the lake's P loads and the associated dramatic decrease in water transparency and oxygen concentrations and increase in algal concentrations. The Québec Ministry of the Environment and Wildlife reported the fish farm's nutrient-rich wastewater added 400 kg of P to the lake annually via the lake's tributary. The wastewater's high nutrient concentration was attributed to the presence of waste food pellets in the effluent.

Lac Heney, located in the Outaouais region of Québec, supports a valuable trout fishery. The lake is deep (30- 33 m maximum depth) with an area of 12.4 km^2. Concentrations of total P in the surface water varied from 10 to 30 µg/L during the summer of 1995. During the period of spring overturn in 1996, P concentrations of 21 µg/L remained uniform in the water column. The concentration of P in Lac Heney classes it as mesotrophic or moderately nutrient rich.

Elevated P concentrations as a result of the fish farm increased the abundance of planktonic algae as well as macrophytes and filamentous algae in the lake (Bird and Mesnage 1996). In 1996, planktonic algal abundance (expressed as concentration of chlorophyll *a*) was fairly low and ranged from 1 to 3 µg/L with concentrations peaking at 14 µg/L in August. The dissolved oxygen profile was typical of a mesotrophic lake whereby concentrations were lower in the deep (hypolimnetic) waters. The result was that the waters of the hypolimnion may have been undesirable for the health of lake trout for most of July because the concentration of oxygen was below 6 mg/L (Prairie 1994). The lowered oxygen levels may also have had a long-term effect by reducing the growth and reproductive rates of sensitive species like lake trout. In addition, the growth of macrophytes was reported to destroy the spawning sites of lake trout.

To create a suitable lake-specific remediation plan for Lac Heney, it was necessary to determine the contribution of P from the fish farm relative to other sources (e.g., atmosphere, cottages, cultivated land in the basin, wastewater inputs, and lake sediments). Sampling undertaken during 1995 and 1996 showed that the quantity of total P entering the lake from all tributaries, including the aquaculture farm was 762 kg (Table 4.2). The aquaculture farm alone contributed 450 kg to this total. The quantity lost from the lake was 231 kg P. The fraction of the contribution that remained in the lake (i.e., the coefficient of retention) was therefore 0.81. Clearly, the fish farm was a major contributor of P to the lake and was responsible for the rapid deterioration in water quality.

Closure of the fish farm is the remediation strategy that the Québec government and the Association for the Protection of Lac Heney has undertaken to reduce P levels and hopefully to avoid further deterioration in the water quality of Lac Heney.

Table 4.2. Calculation of the total phosphorus (TP) balance in Lac Heney, Québec. Adapted from Bird and Mesnage (1996).

Components of the Balance		TP (kg)
Tributaries (excluding fish farm)		312
Aquaculture farm	+	450
Atmosphere	+	186
Cottage septic systems	+	195
TOTAL external contributions	=	1143
Loss of P from the lake	-	231
Quantity of P remaining in the lake	=	912

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clarity and/or to nutrient release from the mussels fecal material (Lange and Wittmeyer 1997; Vogel et al. 1997). However, blue-green algae are not the only algal group that can cause taste and odour problems. Even nutrient-poor lakes and reservoirs may experience intermittent but severe taste and odour, often due to the production of fishy-smelling fatty acid derivatives by chrysophytes (Nicholls 1995; Watson et al. 1999). Public reaction to drinking water described as having an earthy, swampy or fishy taste has led to concern about the safety of the water supply and the implementation of various treatments to mitigate the problem, which have varying degrees of success (Kim et al. 1997; Chow et al. 1998; Satchwill et al. 2000). It should be noted, however, that these odour compounds are not toxic to most organisms, although at excessive levels, geosmin may affect some aquatic juvenile forms (trout, urchins) whereas fatty acid derivatives can be toxic to micrograzers (Nakajima et al. 1996; Gagne et al. 1999; Jüttner 1999).

Algae produce organic matter that forms the base of the food chain, but excessive algal populations have deleterious ecological effects. In the case of nutrient-poor lakes with low production of organic matter, artificial fertilization may be undertaken to provide added nutrients and stimulate plant growth. For example, N and P addition to oligotrophic British Columbia lakes has increased algal and zooplankton abundance and, in turn, increased the size of spawning and juvenile salmon (see Section 3.4). However, in eutrophic lakes, zooplankton go through boom and bust cycles and diversity may be reduced and production limited by blue-green algae blooms. So called "coarse" fish species, such as carp, may come to dominate the fish population while success of spawning and survival of more desirable species, such as trout or bass, becomes more and more tenuous due to the overwhelming amount of organic detritus and periodic de-oxygenation as algal blooms decompose. The result is that eutrophication tends to simplify the aquatic food web. In addition, the overabundance of organic matter caused by excessive algal populations can smother the lake bottom resulting in reduced biodiversity of benthic organisms. Organic matter also accumulates in crevices between rocks where decay can consume enough oxygen to damage the survival of fish eggs. As organic matter accumulates in sediments nuisance accumulations of aquatic rooted plants are more likely to occur. The plants can absorb nutrients from sediments and then release them to the water column either through excretion from the leaves and shoots or by decomposition when the plants die.

The concentration of oxygen in lakes is also related to aquatic plant abundance. Aquatic plants produce oxygen in the daytime as a result of photosynthesis and consume it at night when photosynthesis has ceased and only respiration is occurring. Large populations of aquatic plants can increase oxygen concentrations well above saturation so that the water can effervesce with oxygen bubbles. At night the plants take up oxygen such that oxygen levels may fall well below the saturation concentration (i.e., the maximum concentration of oxygen for its temperature and barometric pressure) thereby causing oxygen stress to fish and invertebrates (Vollenweider 1976). The generally accepted notion that algae cause low oxygen in lakes oversimplifies the situation. Although oxygen may be high during daylight hours, the oxygen decline at night caused by plant uptake can be offset largely by diffusion from the atmosphere into the lake if oxygen concentrations sink below saturation values.

Another way that algae can affect oxygen is by a phenomenon termed "hypolimnetic oxygen depletion". In early summer, solar radiation warms the upper waters of a lake. Although the solar warming extends to a depth of several metres, the deeper waters remain cold. This cold water is denser and therefore does not mix easily with the warmer surface waters. Until the upper layer cools in autumn, the deeper cold water (or hypolimnion) has no direct contact with the air and thus, oxygen consumed there cannot be replaced by diffusion from the air. The rate of oxygen depletion in the hypolimnion depends on the temperature and thickness of the hypolimnion, respiration by bacteria (with the abundance of bacteria determined by the quantity of organic matter settling down from upper warm waters), and respiration by other biota in the hypolimnetic water and sediments (Vollenweider 1976). In lakes severely polluted by raw or treated sewage, the organic matter from sewage plants may be more than sufficient to drive the oxygen consumption. However, in most lakes, excessive algal growth provides the organic matter that bacteria in the lake hypolimnia use for their growth and which, in turn, results in consumption of oxygen (see Lake Erie case study).

Oxygen consumption in a lake's hypolimnion can reduce the oxygen concentration to zero. When this happens, most of the biota are driven from the hypolimnion water and sediment. Those organisms that cannot move fast enough are killed outright. Under less severe circumstances, chronic conditions of low oxygen (5 mg/L) in the hypolimnion can result in a shift in fish composition from cold-water species with requirements for high concentrations of dissolved oxygen (e.g., salmonids and sculpins) to warm-water species (e.g., walleye, pike and smallmouth bass) with lower oxygen requirements. Sometimes, due to wind, the layers slosh around in the lake. Under these conditions there may be fish kills in surface waters when the deoxygenated bottom waters move into lake areas normally overlain by surface water. Oxygen depletion, however, is a natural phenomenon that is exacerbated by anthropogenic pollution. Therefore, the mere occurrence of oxygen depletion in a hypolimnion is not definite proof of a water pollution problem. Rather, the extent and duration of oxygen depletion can be compared to models that take all factors into account to diagnose whether there are excessive sources of organic matter that may indicate pollution sources (Charlton 1980).

Under conditions of low (hypoxia) or no (anoxia) oxygen in the hypolimnion, P can be released from the bottom sediments as phosphate due to the chemical reduction of iron-phosphate complexes. The duration of P regeneration depends on the duration of anoxia and this depends on the amount of organic matter settling down into the hypolimnion. The concern with P regeneration is that it may lead to an in-lake fertilization process (or internal P loading) whereby P from the bottom sediments diffuses upward into the surface water and provides more P for even more algal production. In shallow lakes, decades of pollution can cause release of P from bottom sediments even under oxygenated conditions, slowing the lake's recovery despite reduction of external sources. Water quality in the Bay of Quinte, Lake Ontario responded positively to an 80% reduction in P load despite a strong summertime sediment phosphorus release (Nicholls 1999). Some of the regenerated P is swept from the system by increased river flows in the fall. Thus, internal phosphorus loading may retard but not prevent recovery of lakes that have sufficient flushing.

4.3. Rivers

Until recently, studies on the effects of nutrient addition to aquatic systems focused largely on lakes and reservoirs with comparatively little work conducted on streams and rivers. The relatively little attention originally directed to assessing the impact of nutrient addition on riverine biota likely relates to early beliefs that rivers were conduits for pollutants and could assimilate any loading of nutrients and organic waste. This thought may have been true before the rapid industrialization and acceleration in population growth of this century that resulted in severe degradation of many large rivers (e.g., Thames, England; Rhine, Europe; Elbe, Czechoslovakia/Germany; Cuyahoga, Ohio, U.S.A.). As a result of these observations of gross pollution as well as other observations of increased productivity and/or changes in diversity downstream of point or diffuse sources of nutrient loading, many investigators have examined nutrient dynamics and the response of river communities to nutrient inputs.

Nutrient cycling in rivers involves the uptake and release of N and P by aquatic biota as well as the downstream transport of these elements. In most ecosystems, nutrients largely cycle in place (e.g., between the sediments and water in the case of lakes), and the cycle is often depicted as a circle whereby nutrients move from one compartment (or "pool") to another, eventually returning to the original pool. In rivers, however, this circular process is stretched into a spiral to account for downstream transport. Thus, biological uptake of nutrients during cellular processes, their release during tissue decomposition, and their downstream transport and subsequent uptake by organisms further downstream are referred to as "spiralling" (Webster and Patten 1979; Newbold et al. 1981; Paul and Duthie 1988; Mulholland et al. 1990). The downstream distance that a given phosphate or nitrate ion can travel when it is released in the water column until its uptake and incorporation in biota depends on the degree to which the growth of stream microbes is limited by nutrient availability. Rapid uptake and short transport distances are expected for rivers where plant growth is nutrient limited, whereas N and P will be transported downstream with only minor losses in rivers where factors other than nutrients determine plant abundance.

Besides the amount of nutrients added, the duration of nutrient loading is also important. Pulse events, in which nutrients are added for a limited or defined duration (e.g., a wastewater spill) and are diluted to non-toxic concentrations, often have little effect on an ecosystem in terms of enrichment, and recovery is rapid. Pulse events, however, can be extremely serious when concentrations are sufficiently great so as to be toxic. In contrast, sustained events (e.g., the addition of a continuous effluent discharge for months or years) are generally not toxic but can cause habitat alteration and changes in abundance and composition of riverine plants and animals, depending upon the river nutrient concentrations that result from the sustained event.

Whereas many rivers that have been sites of intensive agriculture and human settlement for many years have become eutrophic or even grossly polluted, rivers receiving moderate additions of nutrients from sewage or agricultural waste (i.e., the pollutant loading is not so severe as to cause deoxygenation) have shown increased biological productivity (see Alberta Northern Rivers case study). Dose-response curves have been developed relating nutrient concentrations to benthic algae ("periphyton") growth. These show that growth saturation of periphyton occurs at very low concentrations, only 1 to 5 µg/L soluble reactive P (Bothwell 1985; Bothwell et al. 1989; Chambers 1996) and 10 to 15 µg/L N (Bothwell 1992). In large rivers, where the dominant primary producers are the phytoplankton, studies have also shown that nutrient addition promotes growth. For example, surveys conducted in southern Ontario and western Québec have found that phytoplankton biomass (expressed as chlorophyll *a* content) is positively correlated with the concentration of total P in the water (Basu and Pick 1996). Thus, small additions of nutrients can result in substantial increases in plant growth.

Increased primary productivity as a result of nutrient loading may, in turn, increase the growth and abundance of higher trophic levels. In the Kuparuk River, a small stream in Alaska, a large increase in periphyton biomass due to P addition increased the growth rate of the four most abundant large insects in the enriched section of the stream (Peterson et al. 1993). Along with the stimulation of invertebrate

Alberta's Northern Rivers: Impacts of Pulp Mills and Municipal Sewage

Expansion of the forestry industry in the boreal forest of western Canada during the past 20 years has raised concerns about the effect of pulp mill effluents on riverine nutrient regimes and the potential for eutrophication. Many pulp mills in western Canada were designed or recently upgraded to address concerns relating to acute toxicity caused by organic contaminants or low dissolved oxygen concentrations. In contrast, the consequences of nutrient loading have not been a major consideration for the management of pulp mill wastes.

The Northern River Basins Study (NRBS) was initiated in 1992 and ran until 1996. The aim of the study was to obtain further scientific information on the existing conditions in, and the effects of development on, the aquatic ecosystems of the Athabasca, Wapiti, Smoky and Peace rivers in northern Alberta (Figure 4.6). The ultimate goal was to provide a basis for wise management of the basins' natural resources. Of the eight major NRBS themes, one focused on assessing the effects of pulp mill and municipal effluents on the nutrient status and potential for eutrophication of the northern rivers. The Athabasca, Wapiti-Smoky and Peace rivers all arise in the mountains. Their basins are sparsely populated relative to their size and largely forested, although agricultural land comprises 17% the Wapiti-Smoky river basin. The waters of the four rivers are intrinsically nutrient-poor: annual median concentrations near the headwaters are < 5 µg/L total P and < 300 µg/L total N.

Figure 4.6. The Athabasca, Wapiti, Smoky and Peace Rivers in northern Alberta.

Research and monitoring studies were undertaken as part of the NRBS to characterize nutrient loading from all point and non-point sources, evaluate the impacts of nutrient loading on river chemistry, assess the response of riverine biota to nutrient loading from pulp mill and municipal effluents *in situ*, quantify nutrient responses of benthic biota, and investigate interactions between nutrients and contaminants in pulp mill effluent on food webs (Chambers 1996, Chambers et al. 2000). Results of these studies showed that elevated N and P concentrations occurred in the Athabasca River downstream of Jasper, Hinton, Whitecourt and Fort McMurray and on the Wapiti River downstream of Grande Prairie during fall, winter and spring. In the Athabasca River, 20% of all total phosphorus (TP) samples and 2% of all total nitrogen (TN) samples collected from 1980 to 1993 exceeded the Alberta Surface Water Quality Objective of 0.05 mg/L TP as P and 1.0 mg/L TN as N. Most of these exceedances occurred during summer and were likely due to high particulate concentrations associated with high river discharge. In the Wapiti River, 74% of TP samples and 19% of TN

Northern Rivers, Alberta concluded

samples collected near the mouth from 1991 to 1993 exceeded the Alberta Surface Water Quality Objectives compared with exceedances of only 12% for TP and 0% for TN upstream of Grande Prairie. This observation suggested that nutrients from the City of Grande Prairie and from Weyerhaeuser of Canada Ltd.'s pulp mill effluents contributed to non-compliance.

Continuously-discharging industrial and municipal sources were found to contribute 4 to 10% of the TN load and 6 to 16% of the TP load in the Athabasca River on an annual basis, with the contribution being higher during winter. Likewise, continuously discharging industrial and municipal sources contributed 20% of the TN and 22% of the TP load in the Wapiti River annually. For the Peace River main stem there was no evidence of nutrient impacts.

Elevated nutrient concentrations in the Athabasca and Wapiti rivers increased periphyton biomass and benthic invertebrate densities and, for the Athabasca River downstream of Hinton, increased the length and body weight of spoonhead sculpin (*Cottus ricei*), a small insectivorous fish species. The increase in periphyton biomass and benthic invertebrate densities downstream of effluent outfalls and, in the case of the benthic invertebrates, no loss of species suggest that the response to effluents is one of nutrient enrichment not toxicity. Studies conducted in artificial streams further showed that periphyton biomass and growth of several mayflies, stoneflies and caddisflies increased in response to nutrient or 1% effluent addition, with no significant difference between the two treatments. These results confirmed that the response to the current level of effluent loading was one of nutrient enrichment. There was no evidence of adverse effects to the ecosystem (Podemski and Culp 1996). Although detailed investigations of spawning grounds and early rearing habitat for fish in the Northern Rivers were not undertaken, it does not appear that dissolved oxygen problems caused by nutrient addition are adversely affecting fish populations at present.

These findings on the contribution of pulp mills to eutrophication of the Athabasca and Wapiti-Smoky rivers resulted in public recommendations to Ministers of Environment in the federal and Alberta provincial governments to eliminate or substantially reduce nutrient discharge to northern rivers, cap direct nutrient loads into specific reaches of the Athabasca and Wapiti-Smoky rivers, and reduce P in pulp mill effluents to minimal concentrations (Northern River Basins Study Board 1996). In the years since these recommendations were made, the Alberta Department of Environment has worked with kraft mill and municipal authorities to decrease nutrient loading to the most affected river (the Wapiti-Smoky system in Alberta) and has stated that any future industrial developments in this basin will be subjected to stringent discharge requirements so as to protect water quality. In addition, the federal government has required all pulp mills to undertake an environmental effects monitoring program that includes an assessment of changes in water quality and biota as a result of effluent inputs.

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growth rates, growth of both young-of-the-year and adult grayling (*Thymallus arcticus*) increased. However, increased benthic invertebrate densities can, in turn, reduce periphyton biomass (Elwood et al. 1981; Winterbourn 1990; Yangdong and Lowe 1995). For example, Biggs and Lowe (1994) found P addition increased the production of invertebrate grazers in the Kakanui River, New Zealand but did not increase periphyton biomass. Removal of grazers resulted in a five-fold increase in periphyton biomass, indicating that a correspondingly more productive, macroinvertebrate community was assimilating increased periphyton production. In contrast with streams, zooplankton are an important part of the invertebrate community in large rivers. Unlike the tight linkage often observed between periphyton and benthic invertebrates in streams, zooplankton biomass in large rivers appears to be only weakly related to phytoplankton biomass (Basu and Pick 1996). This trend may occur because zooplankton are more susceptible to loss than faster-reproducing phytoplankton.

Northern Rivers, Alberta concluded

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The Yamaska River, Québec: a Watershed under Intensive Agriculture

The Yamaska River basin is located in southwestern Québec and stretches from the Appalachian foothills near the U.S. border to the St. Lawrence Lowlands (Figure 4.7). Almost one-fourth of Québec's agriculture takes place in this basin, with hog and corn production being the two leading agricultural commodities. Some 210 000 hectares of land are cultivated (43% of the total area of the basin), mostly in the St. Lawrence Lowlands. Row-crops (mainly corn, but also soybean and vegetables) make up 58% of the cultivated area. Livestock numbers in 1996 were 311 000 animal units (where, for example, 1 animal unit equals 1 cow or 1 horse or 5 hogs between 20 and 100 kg or 125 chickens), including 168 000 hogs, and livestock density was 1.5 animal units per cultivated hectare. Agricultural intensification has increased sharply in the Yamaska basin over the past 25 years so that impacts of agricultural pressures on water resources are now among the highest in the province. Low summer flow (down to as little as 8 m^3/s) places further stress on the Yamaska River and its tributaries (Primeau et al. 1999).

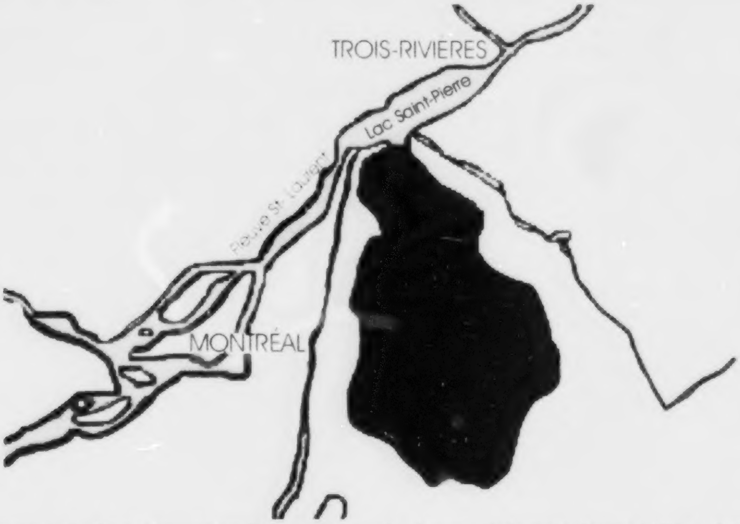

Figure 4.7. The Yamaska River basin, Québec. Adapted from MENV 2000.

Agricultural activities are the principal source of nutrients in the basin. Mineral fertilizers are a source of 13 000 tonnes of N and 3 800 tonnes of P annually, while livestock manure supplies an additional 23 000 tonnes of N and 6 700 tonnes of P. These inputs are very large compared to other sources. Overall, agricultural sources are estimated to contribute 68% of the N and 75% of the P discharged from the river into the St. Lawrence (Delisle et al. 1998). Available nutrients are well in excess of crop requirements in the basin. The Québec Ministry of the Environment estimated recently that organic and mineral fertilizers used in the Yamaska basin exceeded crop requirements for N and P by 144 and 285%, respectively. Excess N and P amount to 80 kg/ha and 27 kg/ha, respectively. Excess nutrients end up in ground and surface waters.

Water quality in the Yamaska River is considered poor by Québec standards. Phosphorus concentrations exceed the Québec guideline for the prevention of eutrophication (0.03 mg/L total P) almost everywhere in the basin. At the mouth of the river, median concentrations for total P (0.195 mg/L) and N (2.15 mg/L) are higher than in any other tributary of the St. Lawrence River (Primeau 1999). Levels of P, N, suspended solids and turbidity throughout the basin are among the highest in Québec and the river is considered to be severely eutrophic.

The most obvious environmental impacts of nutrient enrichment are excessive growth of rooted aquatic plants and algae. In the Yamaska, phytoplankton biomass (expressed as chlorophyll *a* concentration) ranges between 10 and 50 µg/L and increases with distance downstream of the headwaters. These values are very high and provide clear evidence of the advanced state of eutrophication of the river. Large floating mats of Lesser Duckweed (*Lemna minor*) and abundant macrophytes are also observed in reservoirs and backwater zones. The abundance of algae and aquatic plants impair the aesthetic value of the river and its potential for recreational activities.

The Yamaska River, Québec concluded

In addition to causing excessive aquatic plant growth and impairing human enjoyment of the river, eutrophication also affects other aquatic, and sometimes terrestrial, organisms. The degradation of organic matter (macrophytes and algae) leads to oxygen depletion with its adverse, and even lethal, effects on fish and benthic invertebrates. Fish kills have occurred in the Yamaska River in recent years, but exact causes (toxic algae, oxygen depletion, and industrial contaminants) have not been determined. High biological oxygen demand values are observed in the Yamaska River and have been correlated with row-crop acreage, thus suggesting an agricultural origin to oxygen depletion (Primeau 1999).

Larger municipalities in the basin (e.g., Granby, St-Hyacinthe, Cowansville, Farnham) rely upon the Yamaska River as a raw water source for drinking water. Algal blooms cause taste and odour problems necessitating additional water treatment. In addition, high levels of ammonia have been found in river water during warm spells in recent winters, apparently as a result of runoff from agricultural lands. Adequate disinfection of water with high ammonia content can only be achieved through the use of larger quantities of chlorine, thereby resulting in safe but less palatable drinking water.

Water quality in the Yamaska River is considered the worst in Québec. All available data point to agricultural activities as the main cause of the current situation. Better nutrient management, especially of fertilizers (both commercial and manure) is a key factor for the rehabilitation of water and ecosystem quality in the basin. To that end, in 1988 the Québec Government launched a program supporting construction of manure storage tanks. Over 1 million m^3 of manure are now adequately stocked and better managed in the basin. In 1997, the adoption of *Regulations Respecting the Reduction of Pollution from Agricultural Sources* led to a new program supporting agro-environmental initiatives. Central to this strategy is the use of fertilization plans by farmers, whereby fertilizer applications are matched to crop requirements. Recently, the *Union des producteurs agricoles* completed a comprehensive agro-environmental assessment of Québec's agriculture sector, with a view to improving its environmental performance. Clearly, farmers are now more aware of environmental issues, particularly nutrient management, the first step towards establishment of sustainable agriculture in the Yamaska basin and elsewhere in Québec.

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Ministère de l'environnement du Québec

Increases in algal and invertebrate production in response to nutrient addition may, in turn, increase fish production. For example, fertilizer addition to the Keogh River, British Columbia resulted in a five to ten-fold increase in periphyton biomass, which translated to a 1.4 to two-fold increase in salmonid fry weights (Johnston et al. 1990). In the Bow River, Alberta, nutrients from sewage treatment plant discharges sustain an internationally known trout sport fishery in the 50-km reach downstream of Calgary (Culp et al. 1992). Hoyer and Canfield (1991) further showed that total fish standing crop was significantly correlated to total P concentrations for 79 rivers throughout North America.

Because nutrients are responsible for increased growth of riverine plants and changes in the abundance and taxonomic compositions of insects and fish then, in theory, removal or reduction of these nutrient inputs to polluted rivers should improve conditions. This premise is certainly the theory upon which lake remediation rests, namely that reduced nutrient loading (particularly P) will decrease the abundance of planktonic algae and improve water clarity. Compared to lakes and large rivers where water-column concentrations of P and N are good predictors of planktonic algae abundance (Dillon and Rigler 1974; Canfield 1983; Søballe and Kimmel 1987; Basu and Pick 1996), nutrient concentrations show varying degrees of success at predicting periphyton abundance in streams and moderate-size rivers (Biggs and Close 1989; Lohman et al. 1992; Scrimgeour and Chambers 1996; Dodds et al. 1997; Cattaneo et al. 1997; Carr and Chambers 1998; Chételat et al. 1999) and are even

The Yamaska River, Québec concluded

In addition to causing excessive aquatic plant growth and impairing human enjoyment of the river, eutrophication also affects other aquatic, and sometimes terrestrial, organisms. The degradation of organic matter (macrophytes and algae) leads to oxygen depletion with its adverse, and even lethal, effects on fish and benthic invertebrates. Fish kills have occurred in the Yamaska River in recent years, but exact causes (toxic algae, oxygen depletion, and industrial contaminants) have not been determined. High biological oxygen demand values are observed in the Yamaska River and have been correlated with row-crop acreage, thus suggesting an agricultural origin to oxygen depletion (Primeau 1999).

Larger municipalities in the basin (e.g., Granby, St-Hyacinthe, Cowansville, Farnham) rely upon the Yamaska River as a raw water source for drinking water. Algal blooms cause taste and odour problems necessitating additional water treatment. In addition, high levels of ammonia have been found in river water during warm spells in recent winters, apparently as a result of runoff from agricultural lands. Adequate disinfection of water with high ammonia content can only be achieved through the use of larger quantities of chlorine, thereby resulting in safe but less palatable drinking water.

Water quality in the Yamaska River is considered the worst in Québec. All available data point to agricultural activities as the main cause of the current situation. Better nutrient management, especially of fertilizers (both commercial and manure) is a key factor for the rehabilitation of water and ecosystem quality in the basin. To that end, in 1988 the Québec Government launched a program supporting construction of manure storage tanks. Over 1 million m^3 of manure are now adequately stocked and better managed in the basin. In 1997, the adoption of *Regulations Respecting the Reduction of Pollution from Agricultural Sources* led to a new program supporting agro-environmental initiatives. Central to this strategy is the use of fertilization plans by farmers, whereby fertilizer applications are matched to crop requirements. Recently, the *Union des producteurs agricoles* completed a comprehensive agro-environmental assessment of Québec's agriculture sector, with a view to improving its environmental performance. Clearly, farmers are now more aware of environmental issues, particularly nutrient management, the first step towards establishment of sustainable agriculture in the Yamaska basin and elsewhere in Québec.

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Increases in algal and invertebrate production in response to nutrient addition may, in turn, increase fish production. For example, fertilizer addition to the Keogh River, British Columbia resulted in a five to ten-fold increase in periphyton biomass, which translated to a 1.4 to two-fold increase in salmonid fry weights (Johnston et al. 1990). In the Bow River, Alberta, nutrients from sewage treatment plant discharges sustain an internationally known trout sport fishery in the 50-km reach downstream of Calgary (Culp et al. 1992). Hoyer and Canfield (1991) further showed that total fish standing crop was significantly correlated to total P concentrations for 79 rivers throughout North America.

Because nutrients are responsible for increased growth of riverine plants and changes in the abundance and taxonomic compositions of insects and fish then, in theory, removal or reduction of these nutrient inputs to polluted rivers should improve conditions. This premise is certainly the theory upon which lake remediation rests, namely that reduced nutrient loading (particularly P) will decrease the abundance of planktonic algae and improve water clarity. Compared to lakes and large rivers where water-column concentrations of P and N are good predictors of planktonic algae abundance (Dillon and Rigler 1974; Canfield 1983; Søballe and Kimmel 1987; Basu and Pick 1996), nutrient concentrations show varying degrees of success at predicting periphyton abundance in streams and moderate-size rivers (Biggs and Close 1989; Lohman et al. 1992; Scrimgeour and Chambers 1996; Dodds et al. 1997; Cattaneo et al. 1997; Carr and Chambers 1998; Chételat et al. 1999) and are even less successful at predicting the abundance of rooted aquatic plants (Kern-Hansen and Dawson 1978; Canfield and Hoyer 1988; Duarte and Canfield 1990). However, reductions in nutrient loading have improved water quality in some rivers. In the Thompson River, British Columbia, start-up of a bleached kraft mill at Kamloops in 1972 was associated with massive accumulation of benthic algae. Changes to the operation of the Kamloops municipal sewage and the mill wastewater treatment systems resulted in an approximate 35% reduction in total P loading between 1973-75 to 1989. This decline coincided with a 60% decline in periphyton biomass (Bothwell et al. 1992). Similarly, improvements to the City of Calgary's sewage treatment plant reduced total P, total ammonia and nitrate+nitrite loading to the Bow River by 80, 53 and 50%, respectively, between 1982 and 1988. This reduction coincided with a considerable decline in the biomass of rooted aquatic plants (Sosiak 2000).

In summary, studies on the effects of nutrient addition to river systems have generally found that in the case of moderate enrichment to rivers with intrinsically low nutrient concentrations, the productivity of river biota is increased, often without loss of species. Gross pollution involving de-oxygenation of the water results in reduced productivity by periphyton, benthic invertebrates and fish, and loss of species.

4.4. Wetlands

Wetlands are unique ecosystems that are distinguished on the basis of their vegetation, morphology, hydrology and chemistry. They are found in both freshwater and marine environments, and are among the world's most productive environments in terms of carbon incorporation and biomass accrual (Usher and Scarth 1990). Wetlands create habitats for plants and animals: it is estimated that 35% of Canada's rare and endangered animals are in someway dependent on wetlands (Mathias and Moyle 1992). In addition, they perform major ecological functions including provision of nursery or stopping grounds for fish and wildlife, flood control, nutrient retention, purification of aquifers, biogeochemical transformations of elements, and exchange of chemicals, nutrients and organic matter with adjacent ecosystems (Urban and Bayley 1986; Catalla 1993; Hook 1993). Fourteen percent of Canada is covered by wetlands; they are found along the shores of oceans, lakes and rivers, dotted across the Prairies and in countless, poorly drained depressions in the Canadian Shield (NWWG 1988).

The growth and stability of the vegetation in a wetland influences the composition and abundance of invertebrates and vertebrates and defines its biological productivity (Sjörs 1950; Wedeles et al. 1992). Although conversion of wetlands to urban and agricultural land has been the major cause of wetland loss (Moore et al. 1989), a less obvious cause of wetland decline is N addition (Morris 1991; WHO 1997a). Nitrogen often limits growth of wetland vegetation. Excessive inputs may accelerate eutrophication, therefore causing changes in water quality, habitat availability, and biodiversity. Although the effects of nutrient addition on wetlands are less well known than for other aquatic systems, two consequences of nutrient addition on wetland plants are typically observed: an increase in emergent biomass (Willis 1963; Wisheu et al. 1991) and a decrease in plant species diversity (Al-Mufti et al. 1977; Grime 1977; Vermeer and Berendse 1983; Wilson and Keddy 1988). Vegetation surveys and laboratory fertilization experiments suggest that nutrient enrichment of wetlands is a two-stage process. Under moderate nutrient loading, the number of species per unit area and biomass production increase. At higher nutrient loading, plant growth becomes limited due to competition for light (by shoots and leaves) and for space (by roots) (e.g., Willis 1963; Thurston 1969; Grime 1973; Al-Mufti et al. 1977). Plant diversity also gives way to dominance by single species due to invasion by aggressive N-loving, highly-tolerant, non-native vegetation such as purple loosestrife (*Lythrum* *salicaria*), water hyacinth (*Eichornia crassipes*), and giant salvinia (*Salvinia molesta*), or dominance by a monoculture of cattails (*Typha spp.*) or common reeds (*Phragmites spp.*) (Keddy 1988; Mitsch and Gosselink 1993). In forest wetlands, characteristic herbaceous species may be lost and replaced by herbaceous species not normally found in these habitats (Ehrenfeld 1983). These changes in vegetative biomass and species composition as well as associated abiotic changes (e.g., change in water balance due to increased evapotranspiration rates; Chalmers 1979; Howes et al. 1986) can influence the direction of wetland succession.

Changes in agricultural land management practices and sewage inputs have increased the N and P inputs to drainage basins, including wetlands. However, acid deposition is another significant source of N to wetlands (Morris 1991). Not only can the N in atmospheric deposition have direct effects on the abundance and composition of wetland vegetation, it can also change water and soil acidity which, in turn, may affect the availability of nutrients through processes such as phosphate adsorption, nitrification, denitrification and nitrogen fixation, and modify the speciation of metals (Gorham et al. 1984; Crowder 1991; Vaithiyanathan et al. 1996). In addition to direct effects of added N, these alterations may cause serious changes in the entire aquatic food web of wetlands.

Because the quantity of pollutants in water declines as it passes through a wetland, marshes have been considered for their potential to provide secondary treatment of urban and industrial stormwater, sludge and wastewater. However, these effluents also have high BOD concentrations that can lower dissolved oxygen concentrations in wetlands to undesirable levels (Teal and Teal 1969). In addition, nutrient loading from urban waste receiving only primary treatment can overwhelm a wetland and cause it to become choked with phytoplankton, thereby blocking light penetration to deeper vegetation (MacKinnon and Scott 1984). Wetlands are, however, well suited for nutrient removal as part of the "tertiary" or "polishing" stage of sewage treatment and stormwater runoff (MacKinnon and Scott 1984; Kadlec and Knight 1996).

Wetlands in freshwater and marine habitats differ on the basis of their vegetation, hydrology and chemistry. Because of these differences, the effects of nutrient addition will be considered separately for these two systems.

Freshwater Wetlands

Freshwater wetlands exhibit a continuum of variation in space and time relating to characteristics such as the quantity and source of ionic inputs, persistence of standing water, and vegetative composition. At either end of this continuum are ombrotrophic bogs (which are insulated from ground water by a thick layer of peat and receive most of their water and mineral salts from the atmosphere) and swamps and marshes (which have access to mineral-rich ground water). Swamps and marshes (also known as fens in Europe) are usually distinguished by the presence (swamps) or absence (marshes) of trees.

Ombrotrophic bogs are typically acidic. Because they are rain-nourished (i.e., they derive all their nutrients from the atmosphere), ombrotrophic bogs are particularly sensitive to airborne N loads (Urban and Eisenreich 1988). There are no reports as to the effects of N enrichment on ombrotrophic bogs in Canada; however, studies in the Netherlands and the United Kingdom have shown that the dominant species, *Sphagnum*, as well as other plant species favoured by low N concentrations, may decrease in abundance and be replaced by more nitrophilious plants (Lee et al. 1989; Greven 1992; Aaby 1994). Similarly, experimental application of more than 10 kg N (as NH_4/NH_3)/ha/year to Swedish ombrotrophic

Delta Marsh, Manitoba: a Prairie Wetland

Prairie wetlands are complex ecosystems that support a rich and abundant flora and fauna (NWWG 1988). This biodiversity is increasingly threatened by agricultural activities (e.g., nutrient addition, pesticides, plough-down of residues), the ecological effects of which are not fully understood. The fascination with wetlands lies in their rich and diverse wildlife as well as the uniqueness of the landscape (Griffiths and Townsend 1987). For migrating birds, especially waterfowl, wetlands are essential feeding and resting stopover grounds, as well as nesting and moulting grounds (NWWG 1988).

Delta Marsh stretches along the southern shore of Lake Manitoba. It is the largest (200 000 ha) of several marshes in the Lake Manitoba basin and forms one of the finest waterfowl breeding and harvesting areas on the Canadian prairies (MDMNR 1968). The marsh consists of a series of large and small interconnected bays and isolated ponds, separated from Lake Manitoba by a forested sand ridge (MacKenzie 1982). The four dominant emergent species in the Delta Marsh are: *Phragmites australis*; *Scolochloa festucacea*; *Typha glauca*, and *Scirpus lacustris* (Squires and Vander Valk 1992).

Although Delta Marsh borders fertile agricultural land, it is considered naturally eutrophic and not degraded by human activity. The marsh is used as a field station by the University of Manitoba to study the effects of nutrient additions on a prairie wetland. Nutrient additions have been found to enhance primary production in oligotrophic nutrient-poor wetlands in the Interlake Area of Manitoba (Murkin et al. 1992; Murkin et al. 1994; Gabor et al. 1994); however, the effects of nutrient enrichment of a eutrophic prairie marsh are not well understood. Experiments conducted by University of Manitoba scientists and others have shown that N and P addition in the range applied to small heavily-fertilized watersheds (i.e., 2.5 to 10 kg/ha N) can change the species composition of emergent vegetation (Neill 1990). Effects on other organisms (e.g., algae and aquatic invertebrates) are less clear cut, although it appears that increases in nutrient loading, up to a point, are actually favourable to waterfowl by increasing invertebrate production (Murkin et al. 1994). Further research is needed to determine the chemical processes involved in nutrient availability in these systems and the potential effect of fertilization on their productivity and overall function.

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bogs lowered the density of sundew (*Drosera rotundifolia*), a characteristic bog species, due to enhanced competition for light with tall species (e.g., *Eriophorum* and *Andromeda*) which responded positively to the increased N inputs (Redbo-Torstennsson 1994; WHO 1997a). Because increased N loads strongly affect ombrotrophic bogs, due to their high N retention capacity and closed N cycling, WHO (1997a) estimated the critical atmospheric load of N to be 5 to 10 kg N/ha/year for ombrotrophic bogs. Exceedance of this critical atmospheric load would alter the chemical, biological or hydrological function of the wetland.

In contrast to bogs, fens are wetlands that are found in alkaline to slightly acidic habitats (NWWG 1988). Studies of mesotrophic fens in the Netherlands have shown an increase in tall graminoids (grass or *Carex* spp.) and a decrease in the diversity of subordinate plant species in response to N addition (Vermeer 1986; Verhoeven and Schmitz 1991; Koerselman and Verhoeven 1992). Based on these and other Dutch studies, the WHO (1997a) established a critical atmospheric load of 20 to 35 kg N/ha/year for fen systems. Marshes represent wetlands with open nutrient cycles and, as such, are often better equipped to handle the ill effects of nutrient additions (Morris 1991; WHO 1997a; see Delta Marsh case study). However, when the nutrient load, regardless of source, exceeds the filtering capacity of these wetlands, the marshes begin to die (see Cootes Paradise case study).

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Cootes Paradise, Ontario: Urban Impacts on a Great Lake's Marsh

Land use conflicts that pit urban development against maintenance of natural ecosystem function are perhaps best illustrated by the case of wetlands and nowhere is this issue better illustrated than in southern Ontario. Today, the Canadian coastal marshes of the lower Great Lakes remain important resources for waterfowl even though they have undergone drastic change over the last half century from draining, development, eutrophication and pollution (Mitsch 1992).

Cootes Paradise Marsh is a 250 ha wildlife sanctuary located at the western end of Hamilton Harbour, Lake Ontario. It is an important staging area for waterfowl as well as providing nursery and spawning habitat for sportfish, and serving as a local scientific and educational resource (Chow-Fraser and Lukasik 1995).

Over the last century, nutrient inputs from urban, rural and agricultural sources, water level fluctuations, sediment inputs from creeks, and the regular re-suspension of these sediments by wind and the feeding and spawning activities of carp have degraded Cootes Paradise (Chow-Fraser 1998). From 1919 to 1962, a municipal wastewater treatment plant (MWTP) in Dundas, Ontario discharged primary-treated sewage into the western end of the marsh. In 1963, the MWTP was upgraded to a secondary treatment facility (McLarty and Thachuk 1986) although P loading from the MWTP remained high until the early 1970s (approximately 45 kg P/d). In 1978, the plant was again upgraded to a tertiary treatment facility although sand filters to reduce further the loading of P into the marsh were not added until 1987 (Painter et al. 1991). The P loads from the MWTP have steadily decreased from greater than 45 kg/d in the early 1970s (Semkin et al. 1976) to well below 4 kg P/d in recent years (Chow-Fraser et al. 1998). In addition to the MWTP, nutrient loading (especially ammonia and soluble reactive P) has been linked to combined sewer overflows discharging untreated sewage and surface runoff during storm events (Chow-Fraser et al. 1996). Recently, retention tanks have been built to contain effluent from these overflows so that they no longer discharge directly into the marsh (Chow-Fraser et al. 1996). Finally, creeks flowing into the marsh have been contributors of large quantities of sediments and nutrients from the surrounding agricultural land (Chow-Fraser et al. 1996; Chow-Fraser et al. 1998).

The 1978 upgrade of the Dundas MWTP was responsible for dramatic reductions in the concentrations of N and P in the water column (Chow-Fraser et al. 1998; Table 4.3). Prior to the tertiary treatment upgrade, there was a very steep gradient of total phosphorus (TP), soluble reactive phosphorus (SRP), ammonia, and nitrate concentrations from the sewage plant outfall to the marsh outflow. Mean concentrations of TP and SRP entering the marsh at the sewage outfall were almost five and ten-fold higher, respectively, than those in the open water (5 vs. 1 mg P/L TP; 1.4 vs. 0.1 mg P/L SRP); total ammonia-N levels were more than ten times higher (13 vs. 1 mg NH_3^+-N/L) and total nitrate-N five times higher (4.5 vs. 1 mg NO_3^-N/L) (Painter et al. 1989; Chow-Fraser et al. 1998). In the eight years following upgrade to a tertiary facility, the sharp west-to-east gradient in TP concentrations has disappeared (Lougheed and Chow-Fraser 1998). During this period also, the treatment plant began aerating its sewage effluent to reduce the total ammonia-N loading to the marsh, but this additional process has increased nitrate-N loading. Consequently, nitrate concentrations have doubled since 1978 at the sites closest to the sewage outfall (9 to 11 mg NO_3-N/L) and are consistently ten times higher than at the open-water sites (0.5 to 1 mg NO_3-N/L). Moving the sewage outfall to a less restricted area with more mixing could alleviate these conditions.

A common characteristic of many degraded coastal marshes of the Great Lakes is the absence of submersed vegetation. Despite improvements in water quality, less than 10% of Cootes Paradise was covered with either submerged or emergent vegetation by the 1990s (Chow-Fraser et al. 1996) and the prominent members of the emergent community were introduced species, common reed (*Phragmites australis*) and purple loosestrife (*Lythrum salicaria*) (Chow-Fraser et al. 1998). High water levels in the 1940s and 1950s were the original cause for loss of emergent vegetation and resulting habitat changes; however, nutrient enrichment has played a significant role in the degradation of Cootes Paradise (Chow-Fraser et al. 1998).

Cootes Paradise concluded

Table 4.3. Water quality characteristics of Cootes Paradise Marsh in 1973 and 1994 in three stations (open water; vegetated; and Dundas MWTP outfall).

Parameter (mg/L)	**Open water**		**Vegetated**		**MWTP outfall**	
	1973	1994	1973	1994	1973	1994
Dissolved Oxygen - surface	10.5	9.4	NA	7.3	7.4	11.1
Total Phosphorus	2.22	0.13	NA	0.21	15.4	0.15
Total Dissolved Phosphorus	0.96	0.02	NA	0.06	10.7	0.07
Total Nitrate Nitrogen	0.70	0.58	NA	0.35	2.16	10.3
Total Ammonia Nitrogen	0.45	0.14	NA	0.16	8.27	0.27
Total Nitrogen	NA	3.22	NA	2.6	NA	12.6

Data from Bacchus (1974) and Lougheed and Chow-Fraser (1998).

Changes in the algal community of Cootes Paradise over the past four decades have mirrored changes in water quality and plant cover (Chow-Fraser 1998). Although the type of algae found in the marsh was not affected by degrading water quality, the quantity of algae collected at various sites in the marsh was affected (Chow-Fraser et al. 1998). For example, algal counts in an inlet receiving nutrient loading from a storm sewer and runoff from a nearby domestic landfill (Sims 1949) were higher than vegetated sites where poor light penetration and lower nutrient availability contributed to lower algal densities. Blue-green algae numbers increased in the 1970s (Bacchus 1974). However, the introduction of P removal at the Dundas MWTP in 1993 resulted in a reduction in blue-green algal abundance and increased diversity of green algae. There was also a return of *Chrysophyceae* ('golden algae') to the marsh, indicative of less eutrophic conditions (Wetzel 1983; Chow-Fraser et al. 1998). The most significant improvement in terms of the algal community was the development of P limitation in phytoplankton found in the open water.

Vegetation loss in the 1940s caused bottom sediments to become unconsolidated and unable to support a diverse assemblage of aquatic insect larvae (Chow-Fraser 1998). Hence, species diversity of aquatic insects declined dramatically over the past forty years from 57 genera (23 families and 6 orders) in 1948, to 9 genera (6 families and 3 orders) in 1978, to only 5 genera (3 families and 2 orders) in 1995 (Chow-Fraser et al. 1998). The diverse benthic community of the 1940s has now been replaced by a community consisting primarily of chironomid larvae, oligochaetes, and other worms associated with low-oxygen environments (Chow-Fraser et al. 1998). Benthic grazers have been kept in low abundance due to predation by benthivorous carp allowing epiphytic algae to proliferate further, contributing to light limitation of macrophytes.

Historical checklists indicate that Cootes Paradise once supported an important warmwater fishery (Whillans 1996). However, from 1948 to 1978, the abundance of sportfish declined probably in response to high diurnal fluxes in dissolved oxygen (< 2 mg O_2/L at night) in conjunction with the growth of the carp population (Chow-Fraser et al. 1998). The most abundant fish caught in the marsh in recent years have been cyprinids (common carp and hybrids), centrarchids (pumpkinseeds), and clupeids (gizzard shad and alewife), whereas the piscivores such as pike and bass have been exceedingly rare (Kay 1949; Chow-Fraser et al. 1998).

The Hamilton Harbour Remedial Action Plan is currently trying to restore Cootes Paradise Marsh (Rodgers et al. 1992; Chow-Fraser et al. 1996). Nutrient enrichment has been a significant contribution to the degradation of Cootes Paradise. In the absence of nutrient enrichment and other anthropogenic stressors (e.g., further deteriorations in water quality, encroachment by development, and introduction of exotic species), the emergent marsh might be able to recover in future decades.

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Salt Marshes along the Bay of Fundy, Nova Scotia

Salt marshes occur, sometimes extensively, along the coast of the Bay of Fundy, Nova Scotia. These salt marshes are dominated by *Spartina alterniflora* and *Spartina patens*, which grow year-round making their productivity much greater than that of adjacent terrestrial vegetation (Teal and Teal 1969). The marshes in the Bay of Fundy are in many ways unique ecosystems defined by physical processes, of which tidal mixing is most prominent (NWWG 1988; Brylinsky et al. 1996).

Much of the research on *S. alterniflora* salt marshes has emphasized their contribution to marine coastal fisheries (Odum 1980; Pomeroy and Wiegert 1981). The tides that make high production of marsh vegetation possible also remove half the nutrients before the marsh's flora and fauna can use them (MacKinnon and Scott 1984; Keizer and Gordon 1985). In addition, *S. alterniflora* may act as a "pump", taking up P from the sediments and releasing it into tidal waters (Hatcher and Patriquin 1981). Thus, large amounts of nutrients and organic matter are released to adjacent estuaries, stimulating the coastal water food web. In this way, salt marshes enhance coastal fisheries, including shellfisheries (Teal and Teal 1969; Odum 1980).

The flora in some parts of Atlantic coastal plains is found nowhere else in the world (Fernald 1942; Roland 1976) and eutrophication by enriched runoff from land use disturbances could alter the species composition of shoreline vegetation (Ehrenfeld 1983). Coastal plain plants are weak competitors and could be displaced by taller, faster growing invasive plants that would move onto enriched shores (Wisheu and Keddy 1994). Wisheu et al. (1994) recommended, therefore, that in addition to securing habitat, a watershed management plan be developed to prevent nutrient loading from degrading coastal plain habitats.

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Saltwater Wetlands

Saltwater wetlands or "salt marshes" are grass-dominated coastal ecosystems subjected to the inundation of saline tidal waters (Chapman 1974). The salt marsh is the most terrestrial extreme of the marine environment. Plants that live in this habitat are actually terrestrial species that cannot compete with other land plants but are able to tolerate saline conditions (Phleger 1971; Hatcher and Patriquin 1981). Salt marshes are characterized by complex interactions among oceanographic, biological, geological, chemical and hydrological processes (Glooschenko 1979). They are important in the global cycles of N and sulfur (NWWG 1988) by possessing open nutrient cycles and serving as large nutrient reservoirs. In Canada, salt marshes represent a relatively small part of the total wetland acreage (NWWG 1988), in part because of their need for salt water and tides, a narrowly defined physical requirement not commonly found in Canada (Chapman 1960; MacKinnon and Scott 1984).

Salt marshes perform important ecosystem functions including reduction of nutrient and organic export to adjacent coastal waters; spawning grounds for commercially important fish and invertebrates; nesting and staging areas for waterfowl and shorebirds; and areas of waste and toxic substance assimilation (NWWG 1988). In addition, the continuous cycling of nutrients between a salt marsh and the adjacent estuary replenishes the marsh with nutrients, promotes productive mollusc and crustacean growth in the estuary, and enhances offshore fisheries (Teal and Teal 1969).

Salt marshes, in contrast to freshwater wetlands, are strictly N limited (Valiela et al. 1975; Jefferies and Perkins 1977; Cargill and Jefferies 1984; Morris 1991). Information on the impacts of nutrients on saltwater wetlands is not as extensive as for their freshwater counterparts, possibly because the influence of tides that remove some of the nutrients from the systems. However, the effects of

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Salt marshes, in contrast to freshwater wetlands, are strictly N limited (Valiela et al. 1975; Jefferies and Perkins 1977; Cargill and Jefferies 1984; Morris 1991). Information on the impacts of nutrients on saltwater wetlands is not as extensive as for their freshwater counterparts, possibly because the influence of tides that remove some of the nutrients from the systems. However, the effects of

Fraser River Estuary, British Columbia: Marshes versus Development

On the Pacific coast of Canada, estuaries and lagoons make up 10-20% of the shoreline (MacDonald 1977). Coastal marshes are restricted to the head of protected inlets and the deltas of major rivers, with the most extensive being the one that fringes the Fraser River Delta. Here, the chemistry of both the sediments and water is largely determined by the seasonal pattern in discharge and sediment load from the Fraser River (Hutchinson 1982; NWWG 1988).

The southwestern Fraser River Lowland extends from approximately Sea Island east to the Surrey/Langley border and covers most of the land south of the north arm of the Fraser River. This area is one of the most agriculturally productive areas in Canada (Pilon and Kerr 1984). Because many of the wetlands in this region have been drained for agriculture, urbanization, and industrial expansion (NWWG 1988), the wetlands are now largely confined to the 25 213 ha Fraser River Estuary (Ward 1992). Brackish estuarine marshes, characterized by sedges (*Carex lyngbyei*) and bulrushes (*Scirpus americanus*), occur along the delta front where salt and fresh water mix and represent about 93% of the marshes in the estuary. Salt marshes account for the remaining 7%. Productivity of both marsh types is high owing to high nutrient inputs (Bradfield and Porter 1982), especially N (Valiela and Teal 1979).

The amount, location, and nature of urban encroachment to wetlands are subjects of major concern within the Fraser River Lowland (Pilon and Kerr 1984; Ward 1992). Over the last few decades, activities such as dredging, draining and infilling have severely reduced wetland area (NWWG 1988). In addition to overt destruction, construction of dykes and river training structures have had indirect effects on the marshes due to changes in salinity, erosion and sediment redistribution, and nutrient supply (Kerr 1987). For example, activities such as building of causeways and jetties for ship terminals have changed sediment dispersal patterns in some marshes leading to more rapid rates of sedimentation locally and reduced inflow of freshwater (Clague and Luternauer 1982). Although much concern has focused on overt destruction of marsh habitat (Hutchinson 1982), contaminated urban and agricultural runoff, sewage discharge, industrial effluents, landfill leachate and air pollution deposition continue to degrade these wetlands (Harrison 1985; Kerr 1987; NWWG 1988; Stanley Associates Engineering Ltd 1992; Ward 1992; IRC 1994a, b). Perhaps the most detrimental of these sources in terms of high nutrient loads is sewage (Environment Canada and BCELP 1992). Municipal sewage systems discharging to the Fraser River collect and treat not only the domestic flow of over 80% of the approximate 1.5 million people living in the Fraser River Basin population, but also industrial and commercial wastewater (Dorcey and Griggs 1991; UMA 1993). In addition to nutrients, municipal wastewater discharges to the basin contain an array of potentially toxic compounds, including cyanide, chlorine, anionic surfactants, heavy metals, organic materials, phenols and sulphides (Environment Canada and BCELP 1992).

Purple loosestrife recently invaded the mashes of the Fraser River Estuary due, in part, to increased nutrient enrichment (Adams 1993; Grout et al. 1997). Concern has been raised that this plant may be out-competing native plants such as sedges. A shift in the productivity of native plants relative to purple loosestrife in the estuary's vegetation community may not only affect the flora but also the wetland food web due to seasonal differences in detrital supply (Grout et al. 1997). The timing and composition of detrital supplies to the Fraser River Estuary are important factors affecting invertebrate consumers (Sibert 1982). Because purple loosestrife decays more rapidly than sedges, most of the detritus from decomposing loosestrife may be released in the autumn only and not slowly over the whole year (Grout et al. 1997).

The Fraser River Estuary provides homes for the largest population of wintering waterfowl in Canada (NWWG 1988) and is one the most important stopping points for migrating birds on the Pacific Flyway (Cameron and Obee 1981; Leach 1982). It also supports one of the largest salmon runs in Canada (Dorcey et al. 1978; Hall and Yesaki 1987). Maintenance of the wetlands in the Fraser River Estuary will depend upon a balance between development, other human activities and nature.

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increased anthropogenic nutrient inputs, especially N, can have serious consequences. The Bay of Fundy and the Fraser River Estuary on the east and west coasts of Canada, respectively, provide examples of some of these effects (see case studies).

4.5. Coastal Waters

In coastal waters, primary production is largely controlled by nutrient and light availability. The ability to absorb nutrients from the water column, as well as light availability, governs the shift in dominance from slow-growing seagrasses and large macroalgae to faster-growing macroalgae and phytoplankton and, finally, to motile phytoplankton (Duarte 1995). When light is plentiful and nutrients are limiting, the slow-growing seagrasses and large macroalgae are more competitive as they are capable of efficient internal nutrient cycling and, in the case of seagrasses, can access nutrient pools in the sediment (Duarte 1995). As the process of eutrophication continues and nutrient concentrations increase, the abundance of fast-growing macroalgae and phytoplankton increase and, because their biomass occurs close to the water surface, seagrasses are shaded reducing their ability to fix carbon. As the frequency of phytoplankton blooms increases, oxygen production and respiration become uncoupled (Duarte 1995). Increased microbial activity due to sedimentation of organic material causes oxygen demand to exceed oxygen production resulting in deep-water anoxia. Sediment anoxia impairs the ability of seagrasses to acquire N and accelerates seagrass mortality (Duarte 1995). With the loss of seagrasses, turbidity increases as the sediment is no longer stabilized, a situation that further favours motile phytoplankton that can move to the surface to maximize light exposure (Duarte 1995). Anoxia also enhances the release of nutrients from the sediments (i.e., internal nutrient loading), again favouring phytoplankton. This shift in the structure of the marine plant community is a self-accelerating cascade caused by the direct and indirect effects of increased internal nutrient loading and increased shading of benthic plants as the environment shifts from nutrient-limited to light-limited. In the marine environment, this ‘bottom-up’ control of the structure of the primary producer community by nutrients occurs at lower nutrient concentrations than in lakes (Dederen 1992).

The coastal zone is a physically dynamic ecosystem open to constant exchange of water with the open ocean. Circulation and tidal patterns in the near shore and estuarine currents near the mouths of rivers result in mixing of nutrient-poor surface water from the open ocean with the nutrient-rich water of the coastal zone (Persson 1991). The extent of mixing is often determined by salinity and temperature, which set up density gradients that limit blending of different water masses. Typically, as the less dense freshwater enters the coastal zone it creates a net seaward flow of low salinity water on the surface with a net landward flow of deep, denser, saline water (Harrison et al. 1994). This exchange of water is responsible for nutrient dilution along coasts. The extent of these exchanges largely determines the nature and extent of eutrophication in coastal water. In addition to salinity and temperature gradients, deep basins or fjords can also be isolated from water exchange because of physical barriers such as protruding sills (Persson 1991). Depending on the depth of the sill relative to the total basin depth, the deep water in these basins may only naturally re-oxygenate at intervals. The Georgia Basin and Halifax Harbour case studies provide examples of the complex interactions determining nutrient concentrations in coastal waters.

Nitrogen is generally the nutrient limiting primary production in north-temperate oceans, in contrast with freshwaters where P is typically the limiting nutrient (Howarth 1988; Vollenweider 1992). Evidence from the Northern Hemisphere indicates that over-enrichment of coastal waters has created a niche

Marine Aquaculture: Role in Coastal Eutrophication

Nutrient loading from finfish aquaculture operations has the potential to stimulate eutrophication of coastal waters as a result of excretion of dissolved or solid waste, and decomposition of unconsumed feed. The contribution of aquaculture to N and P loading has been identified for several marine operations in Canada. In an assessment of the contribution of salmon aquaculture, traditional fisheries, fish processing plants, a pulp mill and a municipal sewage treatment plant to nutrient loading of Letang Inlet (an embayment of the Bay of Fundy), salmon aquaculture was found to be the largest anthropogenic source of nutrients, contributing 56% of the total N and total P loads to the basin (Strain et al. 1995). These added nutrients could cause eutrophication and possibly trigger microalgal blooms in the vicinity of the cages (Wildish et al. 1990).

In a study of salmonid aquaculture in the Bay of Fundy, localized increases in ammonia were observed close to fish cages although ammonia concentrations for the entire inlet were not above background levels (Wildish et al. 1993). There was no evidence, however, of increased phytoplankton abundance in the vicinities of the cages, which would indicate a eutrophying effect by the salmon cages. This result was surprising as substantial epiphytic communities, including attached macroalgae, generally develop on the pens. In an assessment of the potential for eutrophication from coastal British Columbia fish farms, Black and Forbes (1997) reported that the total dissolved waste from fish farms represented less than 0.3% of the N load to the Strait of Georgia. Taylor et al. (1994) in a study of Sechelt Inlet, British Columbia from 1988 to 1990 concluded that salmon farms could not be shown to affect nutrient regimes or algal blooms in the inlet. Black and Forbes (1997) suggest that the aquaculture industry in coastal British Columbia is unlikely to have a significant widespread eutrophying effect at current production levels.

Although there appears to be no widespread enrichment effects caused by marine aquaculture, there is the potential for local effects immediately underneath cages (DFO 1997). Of the approximately 70% of nutrients typically lost to the environment from aquaculture operations, about 32% of N and 63% of P is lost in particulate form that ends up in the sediment (Holby and Hall 1991; Hall et al. 1992). This influx of particulate organic material can increase oxygen demand in the sediment, thereby increasing sediment nutrient release rates (Holby and Hall 1991; Hall et al. 1992, BCEAO 1997). On the Canadian East Coast, oxygen demand has been found to be four times higher under aquaculture pens than that at reference sites (Hargrave et al. 1993). Mild enrichment by organic sedimentation stimulates the abundance and biomass of the benthic community directly under the pens. If aquaculture pens are poorly sited, pollution sensitive species are lost and the benthic invertebrate community under the cages comes to be dominated by organic pollution taxa, such as the polychaete *Capitella capitata* and nematodes, interspersed among patches of anoxic sediment (BCEAO 1997; DFO 1997). In coastal operations in British Columbia, sediments 20 to 30m away from the pens show no apparent impacts of waste deposition (Cross 1990). Again, proper siting of the pens lessens the impact to the sediments.

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occupied by a diverse group of dinoflagellates and diatoms that like their counterparts in eutrophic lakes, blue-green algae, produce toxic chemicals (Burkholder et al. 1992; Taylor et al. 1994; see PEI case study, Section 5.3). Marine algae have been found responsible for at least four different illnesses in human consumers of molluscs as well as massive mortalities of fish, birds, and marine mammals (Paerl 1997). It is believed that blue-green algae do not dominate primary production in the ocean because of a decreased availability of iron and molybdenum, two micronutrients essential for nitrogen fixation by blue-green algae (Paerl 1997). In the Baltic and Mediterranean Seas, the frequency of blooms of these 'harmful' (i.e., toxic, hypoxia-inducing, food web-altering) algae has increased since the 1950s, as has atmospheric emissions of N associated with urban, industrial, and agricultural expansion in Western and Central Europe (Paerl and Whitall 1999). The occurrence of these 'harmful algal

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Georgia Basin, British Columbia: the Land-Ocean Interface

The Georgia Basin Region is an extensive estuarine system between southern Vancouver Island and the mainland of British Columbia and Washington State (Figure 4.8). It extends from the northern end of the Strait of Georgia to the southern end of Puget Sound and is joined directly to the Pacific Ocean by the Juan de Fuca Strait (West et al. 1994). It represents an area of some of the most rapid population growth in North America. In 1996, approximately 4.5 million people lived in communities surrounding the basin dominated by the cities of Greater Vancouver and Seattle on the mainland and Victoria on Vancouver Island (BCSTATS 1998a; US Census Bureau 1998). By 2010, the population of the area is expected to grow by 28% with much of this growth occurring in unincorporated communities with little to no municipal or intensive regional government (West et al. 1994; BCSTATS 1998b). The potential clearly exists for increased nutrient runoff to coastal waters near these communities.

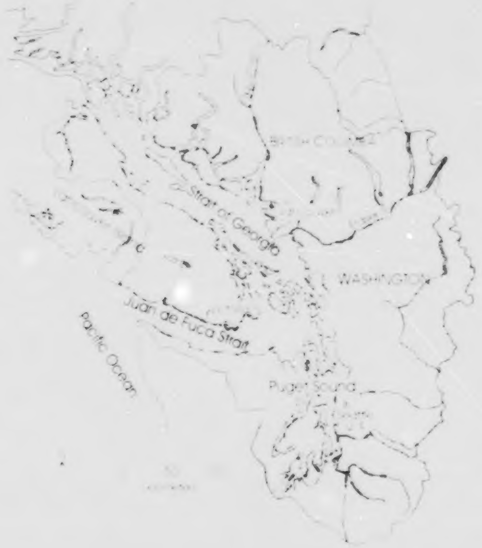

Figure 4.8. The Georgia Basin, British Columbia. The Georgia Basin includes the Straits of Juan de Fuca and Georgia and Puget Sound. Adapted from West et al. 1994.

A net seaward outflow in the upper portion of the water column and a net landward inflow in the lower portion of the water column characterize water circulation in the Georgia Basin (Thomson 1994). Deep inflowing waters supply the Strait of Juan de Fuca with 2 500 to 2 800 t N/d of total N (Harrison et al. 1994; Table 4.4). Depending upon interactions among the general circulation pattern, tides and winds, anywhere from 2 000 to 2 100 t N/d leaves the Juan de Fuca Strait in the seaward movement of surface waters and 500 to 1 000 t N/d is incorporated into biological matter or sediments. Except for a brief period from May to September, natural inputs of nutrients in the Georgia Basin are at concentrations in excess of plant requirements (i.e., nutrient-saturated conditions) (Harrison et al. 1983). This season coincides with the period of maximum algal growth and isolated bays and inlets are most susceptible to the potential impacts of human nutrient inputs. Land-based N inputs to the Georgia Basin from municipal MWTPs discharging directly to the basin plus Fraser River loads were estimated to be 115-130 t N/d (Harrison et al. 1994; Table 4.4). Although P concentrations are very low during the summer, it is rapidly regenerated and, hence, is not limiting (Harrison et al. 1983). Land-based inputs to the Georgia Basin proper are small compared to those cycled naturally as part of the general circulation pattern of the estuary, suggesting that on the broad-scale of the entire estuary, anthropogenic nutrients are not causing eutrophication.

Although natural N inputs to Georgia Basin far exceed anthropogenic inputs, cultural eutrophication has been observed in small, isolated bays and inlets where mixing is reduced. For example, minor increases in phytoplankton abundance and sporadic oxygen depletion have been observed in the bottom waters of Howe Sound, an embayment of Georgia Strait, and have been attributed to nutrient inputs from pulp and paper mills

Georgia Basin concluded

Table 4.4. Estimated nitrogen inputs to Georgia Basin. Adapted from Harrison et al. 1994.

Source	Estimated total N input (t N/d)	
	Most Probable	Upper Limit
Inputs		
Sewage		
Greater Victoria, BC	3.3-3.5	5-6
Greater Vancouver, BC	20-22	
Seattle, WA	15-16	
Total sewage inputs (Victoria, Vancouver, Seattle and other municipalities)	65-70	100
Fraser River (excluding sewage)	50-60	
Total sewage + river	115-130	150-160
Aquaculture	6	
Oceanic deep-water inflow	2500-2800	
Export		
Oceanic surface-water export	2000-2100	

and mine tailings (Harrison et al. 1994). In contrast, it is not clear whether the high abundance of phytoplankton in the south end of Sechelt Inlet is due to the natural shallowness of the area allowing for increased mixing or a mild case of eutrophication due to salmon farms in the area (Harrison et al. 1994).

In recent years, concern has been raised about impact of the discharge of sewage from Victoria following only preliminary treatment (i.e., screening the effluent through 6-mm wire mesh), and, to a lesser extent, Vancouver's sewage effluent. In 1993, Victoria's sewage outfalls located at Clover Point (1 154 m offshore) and Macaulay Point (1 800 m offshore) together discharged approximately 98 000 m^3/d of sewage, representing an N load of 3.3 to 3.5 t N/d (Table 4.4; Taylor et al. 1995). Natural deep-water oceanic N inputs to the Juan de Fuca Strait in the vicinity of Victoria (2 500-2 800 t N/d) exceed sewage inputs by at least 400-fold (Table 4.4; Harrison et al. 1994). Thus, the Clover and Macaulay Point outfalls have little effect on surface water quality in terms of N concentration and hence phytoplankton abundance (Taylor et al. 1995). There is evidence, however, of negative effects of the effluent on benthic invertebrate communities around the outfalls, especially from contaminants (EVS Consultants 1992, Taylor et al. 1995). Similarly, Vancouver's 1992 N inputs from municipal wastewater were estimated to be 21.5 t N/d (Table 4.4; Harrison et al. 1994). Once the natural diffusion of the ocean's deep water into the interior of the Georgia Basin are considered, natural N inputs exceed those of Vancouver's sewage by about 100-fold (Table 4.5; Harrison et al. 1994). Following the1988 extension of the effluent pipe at Vancouver's Iona Island sewage treatment plant to 4 km into the Strait of Georgia in 1988, improvements were observed in Sturgeon Bank. There has been a shift from a benthic algal community dominated by blue-green and green algae to one dominated by diatoms. The density of benthic invertebrates has also increased, as have the oxygen concentrations in overlying waters (Environment Canada 1998a). In 1998, Vancouver upgraded the Annacis and Lulu Island sewage treatment plants to secondary treatment to minimize sewage impacts on the Fraser River and the Georgia Basin in the vicinity of Vancouver.

Currently, it appears eutrophication of the Georgia Basin is not widespread. This situation will likely remain as long as anthropogenic nutrient loading does not increase substantially and regulation of the Fraser River does not reduce flow to the point where estuarine circulation is impaired. Impairment of the natural hydrodynamics that flush the area on a daily basis would likely enhance eutrophication, as anthropogenic nutrient sources would not be washed from the basin.

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Halifax Harbour, Nova Scotia: 200 years of Pollution

Halifax Harbour is a long, irregularly shaped, narrow estuary surrounded by Halifax Regional Municipality, which encompasses the former cities of Halifax and Dartmouth, the town of Bedford, and Halifax County Municipality (Figure 4.9). In 1996, the population of the area was about 343 000 (Halifax Regional Municipality 1998).

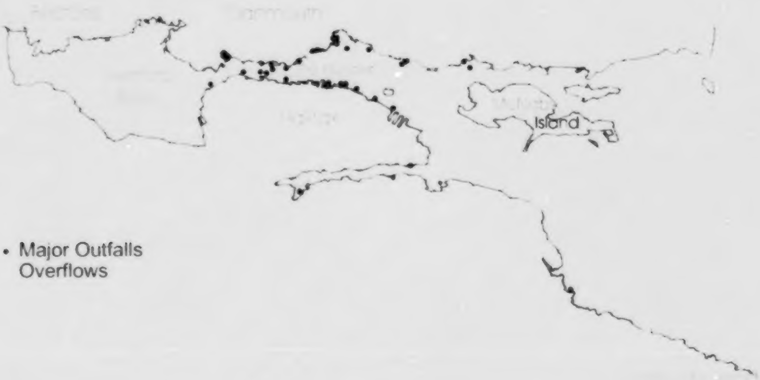

Figure 4.9. Halifax Harbour showing locations of major sewer outfalls and overflows. Adapted from EP-EARP 1993.

Water circulation in Halifax Harbour is typical of estuarine systems with freshwater, primarily from the Sackville River, mixing with surface seawater to create a two-layered flow system (FP-EARP 1993). The upper, less saline water layer (approximately 10m thick) flows toward the Scotian Shelf at rates up to 2.6 cm/s (Petrie and Yeats 1990). In the outer harbour, deep saltier water flows up the inlet toward the head of Bedford Basin at rates of 1-3 cm/s (Petrie and Yeats 1990). The strongest horizontal currents occur in the Narrows and in the McNabs Island area, whereas the most vigorous vertical mixing between the two layers occurs in the Narrows and downtown Harbour area (Petrie and Yeats 1990). The Harbour's two-layer mixing regime is modified by many factors including tides, winds, atmospheric pressure changes, and variations in the volume of freshwater input and rates of vertical mixing (FP-EARP 1993). A sill in the Narrows impedes water exchange within the deep (70 m) Bedford Basin, resulting in naturally higher nutrient concentrations in this area. Consequently, the bottom water of the Basin may become anoxic during years of strong summer stratification (FP-EARP 1993).

Europeans first settled the area around Halifax Harbour approximately 250 years ago and since then, raw sewage and industrial waste have been dumped into its waters. This activity has resulted in a build-up of organic material and metals in the bottom sediments. Currently, sewage is discharged at a rate of 2.1 m^3/s through 20 to 30 outfalls dispersed around the Harbour resulting in a daily load of 0.17 t NO_3-N and 1.6 t P (Table 4.5). This load is slightly larger than NO_3-N load but much larger than the P load from the Sackville River. Taken together the NO_3-N loads from these two inputs are smaller than that contributed by deep oceanic water: however, in the case of P, the Halifax sewage load is similar to the oceanic input (Table 4.5). Petrie and Yeats (1990) concluded sewage makes a negligible contribution to the NO_3-N budget of the harbour but a significant contribution to its P budget.

Halifax Harbour is not a pristine ecosystem. After two centuries of serving as receiving water for domestic and industrial waste, the environment has been contaminated. The complex nature of the system, however, makes it difficult to assess the impacts of added nutrients (DFO/DEMR/DOE 1989). Evidence of eutrophication exists: Bedford Basin experiences periodic fish kills in late summer, presumably due to anoxia from increased oxygen demand associated with decaying organic matter. However, the most recent fish kill in the Harbour (August 1993) was correlated with a dinoflagellate bloom, a class of algae not a normal component of phytoplankton succession in the Harbour. This unusual summer algal bloom may have been stimulated by elevated nutrients brought on by an extended period of calm weather near the end of a drought (P. Strain, DFO, personal communication).

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Halifax Harbour concluded

Table 4.5. Estimated nitrogen and phosphorus inputs to Halifax Harbour. The range in oceanic inflows reflects differences in nutrient concentrations during low and high flows. Surface export was calculated for low flow only.

Source	Estimated Nitrate input (t N/d)	Estimated total P input (t P/d)
Inputs		
Sewage [1]	0.17	1.6
Total river inputs [2]	0.04	0.001
Oceanic inflow [1]	0.53 - 10	0.56 - 2.2
Export		
Surface export	0.78	0.81

[1] Data from Petrie and Yeats (1990)

[2] Data from J. Dalziel, Bedford Institute of Oceanography, unpublished data

Although these events suggest an excess of nutrients in the basin, they may also reflect natural fluxes of nutrients with the tide; however, the waste inputs into the Harbour are likely contributing to the severity, timing, and nature of the fish kills (P. Strain, DFO, personal communication).

Presently, Halifax Regional Municipality is moving toward arresting the discharge of raw sewage into Halifax Harbour. Plans to build a sewage system that would connect to a primary wastewater treatment plant, from which treated effluent would be discharged through one diffuser located west of McNabs Island stalled due to cost (FP-EARP 1993; P. Strain, DFO, personal communication). Concern exists, however, that any decrease in nutrient loading to the Harbour will decrease the organic content of the bottom sediments, thereby eliminating the strongly reducing environment that currently exists and allowing remobilization of heavy metals (DFO/DEMR/DOE 1989).

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blooms' has resulted in closure of shell fisheries resulting in large economic impacts on coastal communities (Paerl and Whitall 1999). Although individuals of the harmful algae species are always present in low concentrations, the exact conditions that cause them to bloom is not clear although blooms do tend to occur following periods of intense rainfall, runoff and intense irradiance (Smayda 1997). For example, laboratory experiments with the diatom *Nitzschia pungens* showed that production of domoic acid, the toxin responsible for amnesiac shellfish poisoning from mussel consumption, was greatest under nutrient (other than nitrate) limitation (Taylor et al. 1994). Another theory currently holds that these blooms may result from changes in the ratio of silica to P or silica to N due to increased terrestrial freshwater runoff (Paerl 1997).

Shifts in the taxonomic composition of the vegetative community also affect the grazer community. Slow-growing seagrasses decompose slowly and experience only moderate losses to grazers. In contrast, phytoplankton production is transferred much faster to heterotrophs, either through increased grazing or decomposition (Duarte 1995). The loss of seagrasses in the coastal zones also means a loss of habitat for many fish and benthic organisms that are important grazers on the phytoplankton. In addition, both anoxia and hypoxia are becoming more prevalent in many estuaries and coastal seas. Increased anoxia in deep water decreases fish egg survival as well as decreasing the area of available habitat for reproduction (Kerr and Ryder 1992). This habitat loss ultimately results in a notable shift from a fish community dominated by the larger, benthic fish species to one composed primarily of smaller, pelagic fish species (Kerr and Ryder 1992).

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Shifts in the taxonomic composition of the vegetative community also affect the grazer community. Slow-growing seagrasses decompose slowly and experience only moderate losses to grazers. In contrast, phytoplankton production is transferred much faster to heterotrophs, either through increased grazing or decomposition (Duarte 1995). The loss of seagrasses in the coastal zones also means a loss of habitat for many fish and benthic organisms that are important grazers on the phytoplankton. In addition, both anoxia and hypoxia are becoming more prevalent in many estuaries and coastal seas. Increased anoxia in deep water decreases fish egg survival as well as decreasing the area of available habitat for reproduction (Kerr and Ryder 1992). This habitat loss ultimately results in a notable shift from a fish community dominated by the larger, benthic fish species to one composed primarily of smaller, pelagic fish species (Kerr and Ryder 1992).

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In the last 20 years, the causes and extent of coastal eutrophication have been increasingly recognized as a global problem (Howarth 1988; Vollenweider 1992; NRC 1993a; United Nations Environment Program 1995; Paerl 1997). Nitrogen addition to coastal waters has increased substantially as a result of human activity: by 25% in the case of N loading from the St. Lawrence River to coastal waters (Howarth et al. 1996). Globally, atmospheric deposition is estimated to contribute approximately 40% of the total loading of new N per year to coastal areas with riverine discharges contributing another 30%, ground water contributing an additional 10% and biological nitrogen fixation adding 20% of new N (Paerl and Whitall 1999). Considering the entire coastline of the North Atlantic Ocean, non-point sources of N are approximately nine-fold greater than inputs from wastewater treatment plants (Howarth et al. 1996). Phosphorus loading has also increased to coastal waters and while non-point source inputs are a significant source of P, point sources may be high in many environments (Caraco 1995; Carpenter et al. 1998). On a more local scale, the expansion of the aquaculture industry has raised concerns about its contribution to eutrophication. Available evidence suggests the potential for local effects immediately underneath cages (see Marine Aquaculture case study).

4.6. Ground water

Nitrate contamination of ground water is usually due to over-application of fertilizer or manure on agricultural land, domestic waste from septic disposal systems, and losses of soil organic N produced by plough down of old meadows, legumes and other crop residues. However, under fertilization of crops can also result in nitrate leaching losses. These losses can be as large as those observed for over fertilized crops because crop growth is impaired leaving a sizeable amount of moisture in the soil. This extra moisture fosters microbial activity resulting in larger net N mineralization (Campbell et al. 1993). Nitrate is highly mobile in soil (Bohn et al. 1985) and, if not assimilated by plants, migrates readily into ground water. Once in ground water, the bacterial reduction of nitrate to nitrous oxide or nitrogen gas is the main potential mechanism for the removal of nitrate. Bacteria capable of denitrification are commonly found in many subsurface environments (Korom 1992). These bacteria require both anaerobic conditions and suitable electron donors such as organic carbon or reduced iron or sulfur for denitrification to occur. The amount of natural organic carbon in subsurface sediments is often very low (Mackay 1990); thus nitrates may persist in ground water due to the absence of a suitable energy source for denitrification. In contrast to nitrate, ammonium is adsorbed onto clay minerals and is not as prone to leaching; however, it is subject to bacterial oxidation to nitrate and may be readily mobilized in aerobic environments. Normally ammonium concentration in soils is very low. Phosphorus movement into ground water is retarded through soil adsorption and mineral precipitation reactions in the unsaturated zone (Wilhelm et al. 1994a; Harman et al. 1996).

Contamination of aquifers with nitrate from nitrogen fertilizers or spreading of animal manure occurs mostly in humid or irrigated areas where sandy soils overlie shallow aquifers (Henry and Meneley 1993). The risk depends on the N input to the land surface and the degree to which an aquifer is vulnerable to nitrate leaching and accumulation. In permeable soils that overlie unconfined, sand or gravel aquifers, leaching of nitrate is likely to occur when the soil moisture capacity exceeds field capacity. Aravena et al. (1993) noted an average concentration of 160 mg/L nitrate-N is typical of shallow ground water in unconfined aquifers overlain by heavily fertilized soils in southern Ontario. However, in clay and clay loam soils that drain slowly after a heavy rain or irrigation application, denitrification can be significant in reducing nitrate in the soils and can prevent leaching to ground water. For example, studies of manure application under dryland farming practices on loam to clay loam soils in central Alberta showed nitrate did accumulate and move down the soil profile under annual manure applications of up to 120 t/ha, but nitrate was not detected in the ground water at depths of 2.5 m or more after several years of study (Chang and Entz 1996, Olson et al. 1997a). Olson et al. (1997a) hypothesized that continued annual application of high rates of manure would cause an accumulation of soil nitrate, with excess nitrate leaching deeper into the soil profile and eventually reaching the water table. Under irrigated conditions, Chang and Entz (1996) found nitrate leached to ground water for annual manure applications of 60 t/ha. No one crop system or tillage method (zero versus conventional tillage) appeared to favour excess nitrate-N accumulation and potential leaching (Sommerfeldt and Chang 1985, Olson et al. 1997b;). A study of feedlots in Alberta found that although levels of N and P were increased in the soil, there was little evidence of extensive downward movement of the nutrients except under one feedlot in a slight depression (Sommerfeldt et al. 1973). Similarly, a lack of increase in soil nitrate concentration beneath a manure storage area in British Columbia was attributed to concentrations of free ammonia in the soil that were toxic to nitrifying bacteria (Zebarth et al. 1999). Extractable soil P concentrations were also higher under the manure storage area. However, instrumentation was not installed to monitor potential leaching of ammonium and P to the ground water at the study site.

Septic systems contribute the largest volume of effluent discharged to ground water in North America (Cherry and Rapaport 1994). Although there are no national statistics on sewage disposal systems in Canada, approximately 25% of the Canadian population is estimated to be served by septic disposal systems. Densities in some areas have increased to the extent that ground water degradation caused by septic systems has become an environmental issue (Harman et al. 1996). Nitrate concentrations at the water table beneath the tile beds of septic disposal systems are roughly 20 to 70 mg/L nitrate-N (Wilhelm et al. 1994b). Septic systems have relied on dilution and dispersion to reduce high concentrations of nitrates. In sand and gravel aquifers and fractured bedrock, low dispersion often maintains high nitrate concentrations for considerable distances (up to several hundred metres) from the drain field. The Cambridge, ON case study provides information about nutrient loading from a domestic septic system.

Nitrogen released from soil organic matter is also a significant source of nitrate for possible leaching to ground waters. Soil organic matter constitutes the main source of N in the soil system. A soil with 5% organic matter will contain about 5 600 kg/ha of N to a depth of 15 cm (Henry and Meneley 1993). Except for intensively managed grasslands (particularly grazed grasslands), perennial grass stands allow very little N movement below the root zone. However, when grass stands are ploughed to return to annual cropping, large leaching losses of nitrate are possible. The presence of bare fallow conditions can significantly increase the amount and rate of nitrate lost below the root zone and potentially add to ground-water nitrate supplies (Henry and Meneley 1993). Leaching, especially of nitrate, can also occur in relatively low-intensity dryland farming where summerfallowing is used. Generally under dryland agriculture, water and nitrate will move below the root zone in early spring when evaporation is low (Stewart 1970). In the Canadian prairies, the high moisture deficit has fostered wide use of summerfallow to ensure water reserves in the soil. During the fallow year in southern Saskatchewan sizeable amounts of nitrate can be leached from the soil, especially in rainy years. In fact, Campbell et al. (1984) estimated that in a year with 23% more precipitation than the long-term average during the growing season, at least 123 kg nitrate-N/ha were leached below 240 cm depth.

Domestic Septic System, Cambridge, Ontario

An intensive study of a contaminant plume from a single-family septic system near Cambridge, Ontario was begun in 1987 by the University of Waterloo and results for nutrient (nitrate and phosphate) contamination at the site have been reported in several papers (e.g. Robertson et al. 1991; Robertson and Cherry 1992; Cherry and Rapaport 1994; Shutter et al. 1994; Wilhelm et al.1994a; Robertson 1995). The site is located on a sand plain where 4 to 8 m of fine to coarse sand overlies a low permeability silt till. Household wastewater is discharged to a holding tank and tile bed that lie at a depth of 0.6 m near the base of the soil zone. The water table is 2 to 2.5 m below the ground surface.

Nitrogen concentrations in the Cambridge effluent average 32 mg/L of ammonium-N and 1.3 mg/L of nitrate-N when it leaves the septic tank (Wilhelm et al. 1994a), values considered typical for wastewater. Between the tile bed and the water table below the tile bed, the ammonium in the effluent is completely oxidized to nitrate in the oxygen-rich unsaturated sand below the tiles. Within the saturated zone, appropriate conditions (i.e. anaerobic environment, carbon source) for removal of nitrate by denitrification are not present; therefore, the nitrate plume persists in the ground water and is very mobile. Furthermore, dilution and dispersion of the nitrate in the plume are inhibited by a background concentration of about 28 mg N/L in the ground water due to the use of agricultural fertilizers on the overlying land. Compared to the N load from agricultural fertilizers around Cambridge, the contribution of nitrate from this septic system to the ground water is relatively small (Wilhelm et al. 1994a).

Phosphate-P levels in the Cambridge effluent averaged 6.3 mg/L for 19 samples collected from 1987 to 1994 and ranged from 1.4 to 14.2 mg/L (Robertson 1995), values also considered typical for domestic wastewater. Approximately 25% of phosphate is attenuated in the unsaturated zone. Attenuation in the unsaturated zone seems to be controlled by phosphate mineral solubility that is established relatively quickly after the effluent enters this zone. Detailed monitoring in the saturated zone has demonstrated the existence of a steadily expanding phosphate plume. Phosphate migration appears to be controlled by sorption processes which retard migration but which provide little further attenuation of the levels that persist. Phosphate is migrating at a rate of about 1 m/yr (Robertson 1995).

The system at Cambridge functions according to design specifications. The average ground-water velocity in the area of the tile beds has been estimated to range from 24 to 40 metres per year. A long narrow plume more than 130 m long and about 10 m wide is defined at the site. A sharp concentration gradient at the edges of the plume implies weak mixing and, therefore, dilution of the plume by the ground water. Shallow domestic wells that lie in the path of a septic system plume may be affected even at significant distance down-gradient from the source (Shutter et al. 1994).

Gary Grove
Environment Canada

Ground water/surface water interactions have not been extensively studied in Canada; therefore, potential impacts from discharge of contaminated ground water are not completely known. However, inputs of N-enriched groundwater to coastal and certain inland waters and P-enriched groundwater to inland waters may contribute to eutrophication. In a study of a septic system plume that had begun to discharge to a nearby river, Robertson et al. (1991) observed almost complete nitrate attenuation within the last two metres of the plume flowpath before discharge to the river. The attenuation was attributed to denitrification occurring within the organic rich riverbed sediments. Several septic system plumes in a barrier sand bar in Point Pelee National Park were investigated for potential interactions with Lake Erie and a large coastal freshwater marsh behind the sand bar (Crowe et al. 1998). Low but measurable concentrations of ammonium and phosphate were measured in the ground water at the edge of the marsh at these sites (Ptacek 1998; Ptacek et al. 1998). However, detailed studies of water

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4.7. Overview of Enrichment in Canadian Ecozones

To assess the geographic extent of eutrophication, information was compiled on water quality of Canadian lakes, river reaches and coastal waters. Review of these reports and papers and consultation with water quality managers showed that regional variation in natural water quality, soil types, climate, landuse, and human impacts resulted in very different impressions of nutrient-related water quality impairment. Even polling of experts at an OECD workshop in 1974 resulted in definitions of lake eutrophy ranging from 38 to 189 mg P/m^3 (mean = 84.4 mg P/m^3) and 861 to 4 081 mg N/m^3 (mean = 1 875 mg N/m^3; Janus and Vollenweider 1981). Most provinces and territories have P or algal guidelines against which to evaluate eutrophication of inland waters; however, there are currently no provincial or national criteria established for N concentrations to control excessive aquatic plant growth in coastal bays and estuaries. Our identification of nutrient-related water quality impairment was based on the trophic status provided by the investigator(s).

British Columbia contains portions of five ecozones; the Pacific Maritime, Montane Cordillera, Boreal Cordillera, Taiga Plains and Boreal Plains. In general, British Columbia enjoys excellent surface water quality with respect to nutrients for fresh and marine waters in all five ecozones. However, the Fraser River Basin, which covers almost one-quarter of the province and straddles the western edge of the Montane Cordillera and the eastern edge of the Pacific Maritime ecozones, shows signs of nutrient enrichment in its lower reaches (Figure 4.10). In this area, agricultural runoff and municipal effluents discharged to the Fraser River and its tributaries (e.g., the Sumas River) combine to create enrichment problems. It has been estimated that 90% of the Province's municipal discharges occur in the Lower Fraser River Basin (Government of Canada 1996). There is little direct evidence of enrichment of coastal marine waters in British Columbia, due to high flushing by nutrient rich water from the open water (see Georgia Basin case study). Only poorly flushed bays show signs of eutrophication, such as low oxygen and high algal biomass.

In the Prairies ecozone of southern Alberta, Saskatchewan and Manitoba, most of the large rivers originate in the Rocky Mountains and flow east to Hudson Bay. As they flow east, the rivers serve critical functions such as power generation, water supply and recreation; all have been regulated. Water quality of the large mountain-fed rivers is generally good, except downstream of large cities and in areas of intensive agriculture (e.g., Old Man River, Alberta; Figure 4.10) (Government of Canada 1996). By comparison, rivers that arise in the Prairies (e.g., Souris River) are usually eutrophic as a result of both natural and agricultural inputs. Lakes in the Prairies ecozone tend to be small, shallow and, in the case of small waterbodies, ephemeral. Nitrogen and phosphorus concentrations tend to be naturally high; intensive agriculture in many watersheds has further elevated nutrient concentrations. A recent program of extensive water quality surveys and intensive research undertaken by the

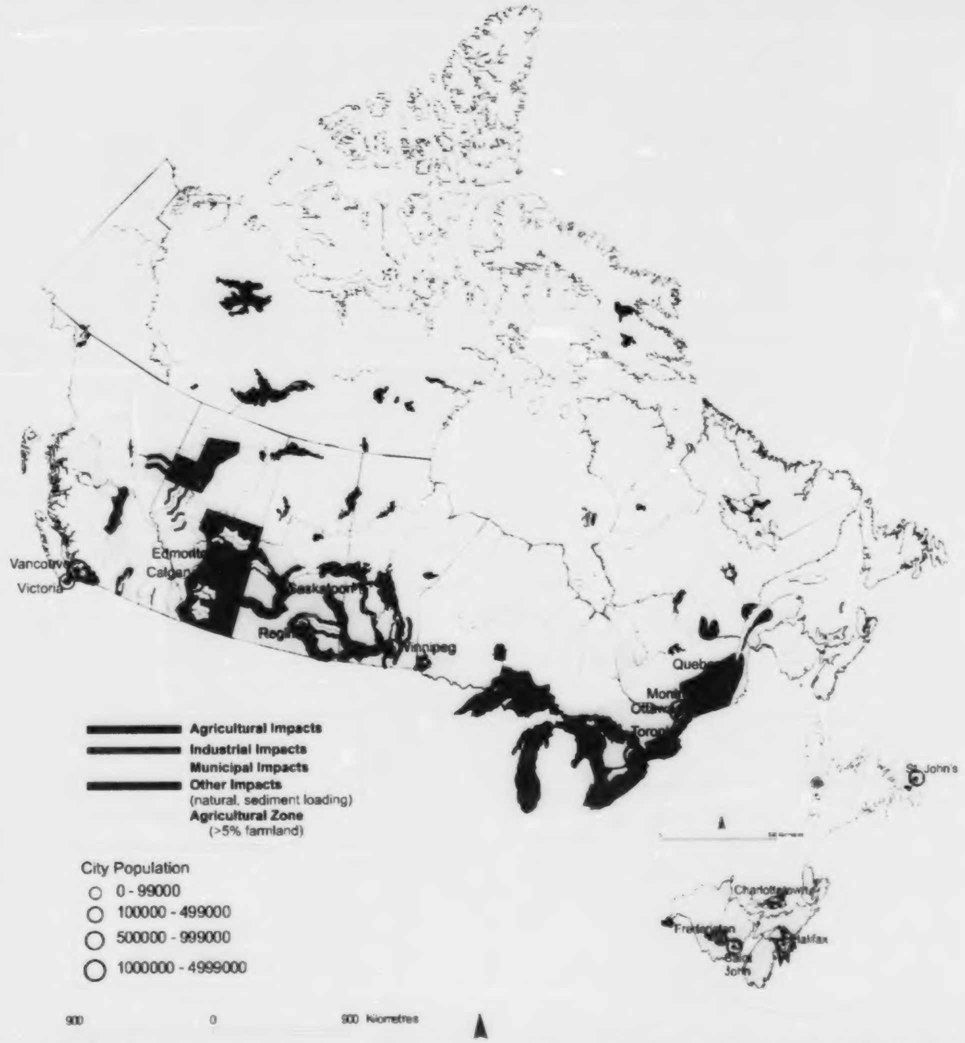
Vancouver
Victoria
Edmonton
Calgary
Saskatoon
Regina
Winnipeg
Quebec
Montreal
Ottawa
Toronto
St. John's
Charlottetown
Fredericton
Halifax
Saint John
Agricultural Impacts
Industrial Impacts
Municipal Impacts
Other Impacts
(natural, sediment loading)
Agricultural Zone
(>5% farmland)
City Population
0 - 99000
100000 - 499000
500000 - 999000
1000000 - 4999000
900
0
900 Kilometres

Figure 4.10. Documented sites of nutrient enrichment in Canada, 1998. Data reflects the both the extent of enrichment and the extent of monitoring undertaken to detect enrichment.

Data Sources:

BC	Government of Canada 1996; BCELP 1999; Harrison et al. 1994; French and Chambers 1995
AB	CAESA 1998; Environment Canada 1998c
SK	Environment Canada 1998c
MB	Environment Canada 1998c
ON	D. Boyd, Ontario Ministry of the Environment, personal communication; A. Gemza, Ontario Ministry of the Environment, personal communication
Great Lakes	Environment Canada 1999c; Government of Canada 1996
QC	Government of Canada 1996; Painchaud 1997
NB	Jerry Choate, NB Environment, personal communication
NS	Darrell Taylor, NS Dept. of the Environment, personal communication; Strain 1998
PEI	Bruce Raymond, PEI Technology and Environment, personal communication
NF	Gerry Collins, NF Dept. of Environment and Labour, personal communication

governments of Alberta and Canada showed that in Alberta, N and P often exceeded interim provincial water quality guidelines for the protection of aquatic life in streams in high and moderate intensity agricultural areas (CAESA 1998). The P guideline was also exceeded in small lakes in high intensity agricultural areas.

The Boreal Shield and Boreal Plains ecozones together cover nearly 2.7 million square kilometres of land and fresh water and include portions of seven Canadian provinces. The rivers and lakes in these ecozones account for 29% of Canada's freshwater surface area (Government of Canada 1996). Overall, water quality is very good. Nutrient concentrations in the eastern portion of the Boreal (i.e., the Boreal Shield) are naturally very low as the underlying rock throughout most of the ecozone is granite and provides little nutrients to overlying soils and water. In the Western Boreal Plains, nutrient concentrations are naturally somewhat higher due to thick glacial soils underlying the rivers and lakes (Cooke and Prepas 1998). Nutrient-caused water quality impairment in the Boreal ecozone is rare and when it does occur, relates to specific anthropogenic stressors (Figure 4.10). For example, rapid development in Ontario's Cottage Country (Haliburton and Muskoka lake districts) during the 1980s resulted in an increase in the concentration of P in many lakes. This increase was largely caused by increased runoff due to land clearing and construction. Since the construction boom has ended and cottage lots have been landscaped, total P values have decreased to values observed pre-development (Andy Gemza, Ontario Ministry of the Environment, personal communication). In 1997, only 20 out of 318 Ontario cottage lakes sampled by the Lakes Partner Program had bays or inlets with total P values greater than 21 µg/L P, the nutrient guideline used in the program to designate them as eutrophic (OME 1998). The majority of these lakes were in the Lake Ontario drainage basin. Similarly, investigations conducted under the auspices of the Northern River Basins Study identified zones of enrichment downstream of specific municipal or industrial outfalls to the Athabasca and Wapiti rivers of northern Alberta (Chambers 1996; Wrona et al. 1996; Chambers et al. 2000).

Agricultural runoff, municipal sewage and industrial wastewater have had a substantial impact on the loading of N and P to rivers and lakes in the Mixedwood Plains ecozone of southern Ontario and Québec. For example, Lake Simcoe and Rice Lake, two large lakes in southern Ontario, have both experienced nutrient impairment due to agricultural and municipal development in their basins (Figure 4.10). The effects of municipal and agriculture development are especially apparent in the Lake Erie basin where some of Canada's most intensive agricultural activity combines with intensive urbanization. Many of the rivers in southwestern Ontario (e.g., the Thames, Grand, Don and Humber Rivers) have total P concentrations that exceed Ontario's provincial water quality objective for rivers of 0.03 mg/L and are thus considered eutrophic (Basu and Pick 1996; D. Boyd, Ontario Ministry of the Environment, personal communication). Although the rivers themselves show varying degrees of excessive plant growth, the cumulative effects of non-point agricultural inputs and municipal and industrial point-source inputs can be seen along the shorelines of the Great Lakes (see Lake Erie case study).

Rivers and lakes in the agricultural basins of the St. Lawrence Lowlands in Québec also reveal the effects of high nutrient inputs. Recent water quality surveys in southern Québec showed that most rivers in the Lowlands had total P concentrations close to the provincial guideline of 0.03 mg/L TP and that agriculture was a major source (Painchaud 1997). In a study of the Yamaska River, 16% of the ammonium-N samples exceeded the guideline of 0.5 mg/L (see the Yamaska River case study). Time-series analysis of water quality data collected between 1988 and 1998 for rivers in southern Québec showed that total P and nitrate concentrations decreased during that period (Simard and Painchaud

2000). This downward trend resulted from the construction of over 500 wastewater treatment plants during the 1980s and 1990s, as well as a reduction in the use of agricultural fertilizers. Although loading from municipal and industrial sources has greatly decreased through the 1990s, agricultural loading remains important. The long-term accumulation of P in agricultural soils as well as the current growth of intensive agricultural operations, such as hog and corn farms, is a source of concern for water quality in the province. Urban sources are still evident in the St. Lawrence River in the form of increased ammonia levels downstream of areas where treated wastewater, storm sewer overflows and raw wastewater are discharged (Hudon and Sylvestre 1998).

In the case of the Great Lakes, concern during the 1960s about the degradation of water quality in the lower Great Lakes (Lakes Erie, Ontario and portions of Lakes Michigan and Huron) led to the signing of the Canada-United States Great Lakes Water Quality Agreement (GLWQA) in 1972. The aim of the GLWQA was to "restore and maintain the chemical, physical, and biological integrity of the waters of the Great Lakes Basin Ecosystem" (Government of Canada 1996). In 1985, the International Joint Commission Great Lakes Water Quality Board identified 42 Areas of Concern (AOCs) around the lakes, 12 of which are in Canadian waters and five are shared with the United States on connecting river systems (Environment Canada 1999c; Figure 4.10). Of the 12 Great Lakes Basin AOCs in Canada, eight have impaired water quality due to excess nutrient concentrations or undesirable algal growth. One of the eight sites is on Lake Superior, two are on Lake Huron, three are on Lake Ontario, and the remainder are on the St. Clair and St. Mary's rivers. All but two are areas where excess nutrients enter harbours from tributaries draining agricultural land and then passing through intensive urban areas. The nutrients mix into and accumulate in the relatively static waters of the harbour. The 1987 revision of GLWQA formalized the establishment of Remedial Action Plans (RAPs) (Environment Canada 1999c). Restoration of beneficial uses within the Areas of Concern (AOCs) is the primary mission of RAPs, and is an essential step in restoring the integrity of the Great Lakes Basin ecosystem. The RAP program emphasizes adoption of a systematic and comprehensive ecosystem approach to restoring and protecting beneficial uses and the area's ability to support aquatic life (Environment Canada 1999c). This process has resulted in decreased nutrient loading from tributaries, although in some cases P concentrations remain high due to internal loading from harbour sediments and/or insufficient reduction in nutrient loads. Successes have been achieved through the RAP program with one AOC previously identified as having nutrient impairment, Collingwood Harbour, now considered restored (Environment Canada 1999c).

The Atlantic ecozones include the terrestrial Atlantic Maritime ecozone and the marine Atlantic and Northwest Atlantic ecozones and encompass the provinces of Nova Scotia, New Brunswick, and Prince Edward Island. This area of Canada has been subjected to intensive alteration in its almost 400 years of European settlement. For many of the large cities, untreated sewage has been discharged to coastal waters for over 250 years. Although this long history of untreated sewage discharge has caused a variety of problems, the complex nature of the coastal environments, high flushing rates and inputs of nitrogen-rich deep oceanic waters make the impacts of anthropogenic nutrients difficult to quantify. A recent assessment of water quality data collected from 34 Maritime inlets showed that the harbours can be arranged along a nutrient gradient; however, this gradient likely relates to the N and P content of the incoming ocean water rather than anthropogenic inputs (Strain 1998). Inland waters throughout the region have been affected by agriculture and industrial development and show varying degrees of eutrophication (e.g., Saint John River, NB; Boughten River, PEI; Black Brook, NB; Figure 4.10).

4.8. Conclusions

Changes in species abundance and dominance due to nutrient addition from human activities are a pervasive problem in Canadian ecosystems, particularly for those systems that are sites of intensive settlement, agriculture or industrial activity.

Fertilization studies involving single N applications have confirmed that N is the major limiting element for tree growth in many Canadian forest ecosystems. In recent decades, atmospheric deposition of N has been found to have steadily increased in parts of Canada and, because most of Canada's forests are N limited, this increased N deposition has likely increased the productivity of forest vegetation (Table 4.6). Controversy exists, however, as to whether the productivity gains resulting from increased N deposition along with a warming climate and elevated CO_2 levels in the atmosphere have offset the carbon loss caused by timber harvest, fire and insect-induced mortality. Atmospheric deposition of nitrate has also resulted in the forests surrounding Turkey Lake in central Ontario and Lac Laflamme in southern Québec being in an early stage of N saturation (Jeffries 1995).

Lakes in developed watersheds often have elevated nutrient concentrations, which increase the abundance of planktonic and sometimes filamentous algae and, in turn, change the abundance and composition of invertebrates and ultimately fish (Table 4.6). The increased organic matter production resulting from nutrient enrichment can also increase the abundance of bacteria, and their metabolic activity can lower oxygen concentrations in the water to the extent that fish survival is threatened. Although these effects will occur throughout a lake, they will be intensified near the nutrient source, e.g. near sewage or municipal wastewater discharges or near tributary inflows. Similar enrichment effects have been observed for rivers draining populated or agriculturally intensive lands. Here river volume, and thus the extent to which the pollutant is diluted, largely determine the severity of nutrient addition. Moreover, in rivers, effects of nutrient addition may be compounded when one input after another is added to the river before the effects of more upstream inputs are attenuated. This accumulation of nutrients is of particular concern for northward flowing rivers where settlement and industrial development tend to occur in the southern headwaters and pollutants can thus travel hundreds of kilometres through terrain that has otherwise experienced minimal human impacts. However, it should be noted that the changes in the food web that occur in response to nutrient addition are not always perceived to be negative. For example, nutrient-poor coastal lakes and rivers in British Columbia are purposely fertilized to enhance the food supply and thereby increase the size of juvenile and spawning salmon.

Marine and freshwater wetlands in Canada also show evidence of enrichment. Although the major cause of wetland loss is their conversion to agricultural or urban land, nutrient addition can cause changes in the composition of emergent vegetation and, thereby, affect the quality of water transferred to adjacent ponds, lakes and coastal waters (Table 4.6).

Coastal waters on both the Pacific and Atlantic shores of Canada have had a long history as receiving waters of untreated sewage. Although a number of environmental issues have been associated with marine discharge of untreated sewage, the complex nature of coastal environments, high flushing rates and inputs of nitrogen-rich deep oceanic waters have generally minimized any widespread effects of anthropogenic nutrients. Certain small bays and harbours where oceanic exchange is limited, do however, show evidence of eutrophication.

Table 4.6. Nutrient enrichment: a summary.

Potential enrichment effects on:			
Water/soil quality and characteristics	**Plants**	**Animals**	**Human health and economy**
Forests (Section 4.1)			
• N saturation is associated with changes in soil chemistry, such as loss of nutrient cations, an increase in soil acidity, and an increase in the availability of aluminum	• increased biomass of trees and other vegetation in forests • ultimately, a decline in forests if soil quality deteriorates		
Lakes (Section 4.2; Lake Erie, Qu'Appelle Lakes and Lac Heney case studies)			
• elevated nutrient concentrations • reduced water clarity • depletion of oxygen	• increased phytoplankton, filamentous algae and/or aquatic weeds • increased abundance of potentially toxic blue-green algae	• increased productivity of benthic organisms and fish at moderate nutrient levels • at high nutrient levels, reduced biodiversity of benthic organisms due to lowered oxygen; reduced survival of fish eggs due to oxygen depletion; changes in biodiversity of fish and benthic organisms.	• degraded shorelines and reduced recreational use due to nuisance algae/weeds; • loss or impairment of fisheries • increased costs for drinking water filtration; taste/ odour problems; blockage of water intakes • toxins produced by blue-green algae can poison livestock and people
Rivers (Section 4.3 and 5.2; Alberta's Northern rivers and Yamaska River case studies)			
• elevated nutrients concentrations • reduction in oxygen concentrations	• increased benthic algae, filamentous algae, planktonic algae and/or aquatic weeds	• increased productivity of higher trophic levels (e.g., insects and fish) at moderate nutrient levels • reduced productivity by benthic organisms and fish and loss of species at high nutrient levels.	• reduced recreational use due to nuisance algae/weeds; • increased costs for drinking water filtration taste/ odour problems; blockage of water intakes
Freshwater Wetlands (Section 4.4 and 5.2; Delta Marsh and Cootes Paradise case studies			
• elevated nutrient concentrations • water balance changes due to increased evapotranspiration rates caused by changes in plant biodiversity.	• bogs: decline in *Sphagnum* and replacement with more nitrophilous mosses; • fens: increase in tall graminoids (grasses, sedges) and loss of shorter plant species that are less able to compete for light; • marshes: increase in phytoplankton and emergent vegetation	• bogs, fens: changes in wildlife and waterfowl biodiversity if changes in plant biodiversity cause loss of habitat or food supply	

Table 4.6. Nutrient enrichment: a summary (concluded).

Potential enrichment effects on:			
Water/soil quality and characteristics	**Plants**	**Animals**	**Human health and economy**
Salt Marshes (Section 4.4; Bay of Fundy and Fraser River Estuary case studies)			
• elevated nutrient concentrations • water balance changes due to increased evapotranspiration rates caused by changes in plant biodiversity.	• increase in phytoplankton that block light penetration to deeper vegetation. • loss of plant diversity	• nutrients released from salt marshes to adjacent estuaries may stimulate coastal water food webs. • changes in wildlife and waterfowl biodiversity if changes in plant biodiversity cause loss of habitat or food supply	
Coastal waters (Section 4.5; Georgia Basin and Halifax Harbour case studies)			
• increased nutrient concentrations • deep-water anoxia due to increased organic loading • turbidity due to loss of seagrasses to stabilize bottom sediments	• increase in phytoplankton and macro-algae • increased abundance of algae that produce toxic chemicals. • reduction in seagrasses due to reduced light penetration and sediment anoxia	• loss of habitat for fish and benthic organisms	• four different illness in human consumers of molluscs; • large economic impacts on coastal communities as a result of closures of shellfish fisheries;.
Ground water (Section 4.6; Domestic Septic System case study)			
• increased nitrate and P concentrations in certain aquifers			• nitrate contamination of aquifers used as drinking water source

Elevated nutrient concentrations, particularly nitrate, are also found in ground waters (Table 4.6). Aquifers, which are shallow (<30 below the surface) and situated in very sandy, porous soil, are particularly vulnerable to nitrate contamination. Nitrate contamination of ground water is a concern to consumers of this drinking water source and may also contribute to the eutrophication of inland and coastal waters.

Nutrient addition and its consequent effects on biodiversity are found across Canada, wherever intensive agriculture, urbanization or industrialization occur. Areas of concern with respect to nutrient addition are the forests, lakes and rivers of the Lower Fraser Valley; the surface waters of the southern Prairie Provinces; the rivers draining into Lake Erie, Lake Ontario and the St. Lawrence River; and rivers draining agricultural watersheds in the Atlantic Provinces and the estuaries and coastal waters into which these rivers flow.

5.0 Direct and Indirect Toxic Effects of Nutrient Addition

Highlights

- Toxic forms of nitrogen include un-ionized ammonia (NH_3), ammonium (NH_4^+), nitrate (NO_3^-), nitrite (NO_2^-), and nitrogen dioxide (NO_2). Phosphorus, in the forms present in the environment, is not toxic even at high concentrations.

- Nitrates from agriculture have contributed to the recent decline in frogs and salamanders in Canada. Concentrations of nitrate greater than 60 mg/L in water kill the larvae of many amphibians.

- Elevated nitrate concentrations have been shown to stimulate the growth of a marine diatom capable of producing domoic acid, an algal toxin. In 1987, the PEI mussel fishery was closed for the remainder of the season after mussels accumulated domoic acid to levels toxic to humans.

- 26% of Canadians, approximately 8 million people, rely on ground water for domestic water supply. In most provinces, 5-20% of these wells are contaminated with unsafe levels of nitrate with up to 60% of wells contaminated in regions with high-demand crops or intensive livestock operations. Agriculture and septic beds are the largest human-derived contributors to ground water contamination in Canada.

- Methaemoglobinaemia or "Blue Baby Syndrome" occurs when nitrite (NO_2^-) reacts with the iron in haemoglobin to reduce the oxygen-carrying capacity of haemoglobin.

- Use of nitrogen fertilizers, including manure, in agriculture has resulted in incidences of nitrate poisoning in livestock. Feed crops accumulate nitrate while growing which can be passed on to livestock during feeding. The consequences of nitrate poisoning to livestock include decreased milk production, weight gain, and disease resistance.

Nitrogen and phosphorus are essential elements for all organisms. P is not directly toxic in the forms and amounts found naturally in the environment (Switzer-Howse and Coote 1984). However, elemental phosphorus (P_4) is known to be toxic to humans and other animals (Isom 1960; Idler et al. 1981). Elemental P does not occur naturally. It is produced by combining phosphate-bearing rocks, coke and silica in an electric arc furnace and is used in the production of baking powder, tooth paste, dyestuffs, pharmaceuticals, fertilizers and dozens of other products. Because elemental P quickly reacts with oxygen when dissolved in water to form non-toxic phosphate, dilution of any accidental spill of elemental P results in its rapid disappearance (Jangaard 1972); thus, cases of elemental P toxicity in Canada are extremely rare (e.g., Jangaard 1972; Idler et al. 1981). In contrast, high concentrations of certain N forms can cause adverse health effects for plants and animals including humans. This chapter describes nitrogen toxicity for aquatic and terrestrial plants and animals, including humans, and then evaluates the extent to which toxic concentrations of N and their impacts on biota occur in Canada. The role of N and P in contributing to the development of toxic algal blooms is also considered. In addition to the information in this chapter, Environment Canada is undertaking an assessment of ammonia toxicity as part of a review of compounds for inclusion in the Federal Government's *Canadian Environmental Protection Act* Priority Substances List (see *CEPA* PSL2 text box).

Canadian Environmental Protection Act Priority Substances List II: Problem formulation for ammonia in the aquatic environment

In 1995, the Ministers' Expert Advisory Panel listed ammonia in the aquatic environment as a compound that should be placed on the second *Canadian Environmental Protection Act* (*CEPA*) Priority Substances List. The rationale for its inclusion was: "Anthropogenic sources of ammonia in Canada include effluent from sewage treatment plants, steel mills, fertilizer plants, the petroleum industry, and intensive farming; and releases from the sources can result in local elevated concentrations" (Environment Canada 1997b). As a result, an assessment of ammonia in the aquatic, terrestrial and atmospheric environments was initiated by Environment Canada to evaluate the extent of the problem in Canada.

The *CEPA* assessment identifies the release and use of ammonia into the Canadian environment (Environment Canada 1999b). In addition, it assesses the fate of ammonia in the atmosphere, soil, groundwater, surface water, and sediments. Aquatic toxicity to plants, microbes, invertebrates, and vertebrates and the ecological relevance of the findings are also examined.

Preliminary findings from the *CEPA* assessment indicate that:

- the major, quantifiable, source of ammonia to aquatic ecosystems in Canada is municipal sewage effluents;
- the major industrial source of ammonia to the Canadian environment is the fertilizer industry and most of its releases are to the air;
- agricultural operations, specifically intensive livestock rearing, are a large source of ammonia to air and soil; however, contributions to water are difficult to quantify;
- further study is required to quantify accurately ammonia releases from automobile emissions.

Releases of ammonia to freshwaters have the potential to affect many aquatic organisms deleteriously. Preliminary results indicate that the acute critical toxicity value for freshwater organisms is 0.29 mg/L un-ionized ammonia, based on a lethal response of 37 freshwater species (including fish, aquatic insects, aquatic worms, crustaceans and clams). The chronic toxicity value for freshwater fish was 0.041 mg/L un-ionized ammonia, based on a 5% decline in growth/reproduction of seven fish species, an amphibian species and five aquatic invertebrate species. Due to the lack of recent marine toxicity data, it was decided to use the U.S. EPA's Genus Mean Acute Toxicity Value of 0.49 mg/L for winter flounder. A chronic toxicity value for marine species is not available (Environment Canada 1999b).

The *CEPA* assessment also noted that ammonia gas in Canadian air is generally non-toxic, except for some accidental industrial releases. Although ammonia deposition from the atmosphere onto forests, fens and moors in industrialized regions of Europe has caused terrestrial eutrophication, this phenomenon has not yet been documented as happening in Canada. There are, however, potential instances of this occurring in Canada, for example in the Lower Fraser Valley and in Ontario just north of Toronto (Environment Canada 1999b).

Environment Canada is currently completing its assessment of ammonia in the Canadian environment and Health Canada is reviewing the health-related effects from exposure to ammonia in ambient air. These technical documents will form the basis for a decision as to whether ammonia should be added to the *CEPA* List of Toxic Substances. If ammonia is added to the list, then new management policies may be implemented for the release of ammonia to aquatic environments.

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5.1. Laboratory Studies of Toxic Effects of Nitrogen

Aquatic plants

Nitrogen is essential for plant growth. It is a component of protein, chlorophyll and other organic compounds. For most plants, N can be assimilated as ammonium and nitrate; however, although ammonia is an important nutrient, it is toxic at higher concentrations.

To appreciate the toxicity of ammonia, a basic understanding of its chemical forms is necessary. In aqueous solutions, a chemical equilibrium is established between un-ionized ammonia (NH_3) and the ammonium ion (NH_4^+):

$$NH_3 + H_2O \rightleftharpoons NH_4^+ + OH^-$$

The total ammonia content of a solution is the sum of un-ionized ammonia (NH_3) and the ammonium ion (NH_4^+). The proportion of un-ionized ammonia in a solution is related to both the temperature and pH of the solution such that the proportion of un-ionized ammonia is greater at high temperature and high pH (i.e., alkaline solutions). Concentrations of ammonia that are toxic for aquatic animals are generally expressed as un-ionized ammonia, because un-ionized ammonia and not the ammonium ion has been found to be the toxic form (Mueller and Helsel 1996).

Data on the toxicity of ammonia for freshwater phytoplankton (e.g., Abelovich and Azov 1976; Bretthauer 1978) and vascular plant communities (Stanley 1974; Welch 1992) are limited. Results suggest that freshwater plants are more tolerant to NH_3 than invertebrates or fish (NRC 1979; Environment Canada 1997b) and that toxicity is probably due to un-ionized ammonia (NH_3) rather than ammonium (NH_4^+) (WHO 1986). The few studies available generally show that ammonia at concentrations > 2.4 mg NH_3/L inhibits photosynthesis and growth of algae (Abelovich and Azov 1976; WHO 1986). Experiments with rooted aquatic plants have also reported ammonia reduced the length and weight of roots and shoots (Stanley 1974; Litav and Lehrer 1978). Ammonia has also been investigated for its potential as an aquatic herbicide. Complete eradication of rooted aquatic plant species (water shield, *Brasenia schreberi* and American lotus, *Nelumbo* sp.) was achieved with ammonia treatment of 25.6 mg NH_3/L (Champ et al. 1973). These authors also noted a 95% reduction in phytoplankton density (dinoflagellates, diatoms, desmids, and blue-green algae) two weeks after treating a small natural pond with 3.6 mg/L ammonia (NH_3).

There have been no conclusive toxicity tests on saltwater plants. *Coccolithus huxleyi* was found to be the most sensitive Chrysophyte to ammonium sulfate (Pinter and Provasoli 1963). Ammonium sulfate was also found to be toxic to some marine dinoflagellates, but only at ammonium concentrations far exceeding those in sea water (Provasoli and McLaughlin 1963). The euryhaline alga *Pyrmnesium parvum* was effectively controlled by 100 mg N /L as ammonium sulfate as a result of the lytic effect of un-ionized ammonia (Shilo and Shilo 1953; 1955). A concentration of 0.24 mg NH_3/L retarded the growth of seven of ten species of benthic diatoms cultured for 10 days by Admiraal (1977). In a study of ammonium-induced production of the marine phycotoxin, domoic acid, in the diatom *Nitzschia pungens*, Bates et al. (1993) observed enhanced domoic acid production at concentrations of 7.92 mg NH_4^+/L, whereas concentrations of 15.84 mg NH_4^+/L prevented the growth of diatoms. Decreased cell yield and inhibition of photosynthesis were also associated with increased domoic acid output. The authors suggested that enhanced production of domoic acid is a response to ammonia toxicity.

Table 5.1. Acute effects of un-ionized ammonia on aquatic animals. All endpoints were LC_{50} (the concentration of NH_3 at which 50% of the test organisms died.)

Organism		Life Stage	Conc. ($mg\ NH_3/L$)	Exposure period	Reference
Vertebrates					
Acipenser brevirostrum	Shortnose sturgeon	fingerlings	0.58	96 h	Fontenot et al. 1998
Catastomus commersoni	White sucker		1.53	96 h	Arthur et al. 1987
Chasmites brevirostris	Shortnose sucker	larvae	1.24	96 h	Saiki et al. 1999
		juvenile	0.48	96 h	Saiki et al. 1999
Deltistes luxatus	Lost River sucker	larvae	0.51	96 h	Saiki et al. 1999
		juvenile	0.92	96 h	Saiki et al. 1999
Diplodus sargus	Porgy	larvae	0.36	24 h	Brownell 1980
Etheostoma spectabile	Orange throat darter		0.74-0.88	96 h	Hazel et al. 1979
Gaidropsarus capnesis	Cod larvae		0.46	24 h	Brownell 1980
Gasterosteus aculeatus	Threespine stickleback		0.71-0.84	96 h	Hazel et al. 1971
Ictalurus punctatus	Channel catfish	fingerlings	2.36	24 h	Robinette 1976
		juveniles	1.6-3.1	96 h	Colt and Tchobanoglous 1976
			2.4-3.8	96 h	Colt and Tchobanoglous 1976
			0.27	24 h	Spehar et al. 1981
			0.86	96 h	Arthur et al. 1987
Lepomis cyaneilus	Green sunfish	juveniles	0.5-1.73	96 h	McCormick et al. 1984
Lepomis macrochirus	Bluegill		1.17	24 h	Hazel et al. 1979
Lithognathus mormyrus	Mormora	larvae	0.38	24 h	Brownell 1980
Micropterus dolomieui	Smallmouth bass	juveniles	0.69-1.78	96 h	Broderius et al. 1985
M. treculi	Guadelupe bass	fingerlings	0.56	96 h	Tomasso and Carmichael 1986
Morone saxatillis	Striped bass		0.76-1.2	96 h	Hazel et al. 1971
			1.01	96 h	Oppenborn and Goudie 1993
Notropis lutrensis	Red shiner		2.3-2.6	96 h	Hazel et al. 1979
			0.99	96 h	Rosage et al. 1979
Oncorhynchus gorbuscha	Pink salmon	eggs & larvae	0.083	96 h	Rice and Bailey 1980
O. kisutch	Coho salmon	fingerlings	0.28-0.59	96 h	Robinson-Wilson & Seim 1975
		fingerlings	0.51	48 h	Buckley 1978
		fingerlings	0.45	96 h	Buckley 1978
O. mormyrus	Chinook salmon	parr	0.87-1.0	24 h	Harader and Allen 1983
O. mykiss	Rainbow trout	fry	0.072	24 h	Rice and Stokes 1975
			0.3-0.8	96 h	Thurston et al. 1981
			0.16-0.50	96 h	Thurston et al. 1981
			0.53	96 h	Arthur et al. 1987
O. nerka	Sockeye salmon	eggs	0.11	62 days	Spehar et al. 1981
O. tshawytscha	Chinook salmon	parr	0.36	96 h	Harader and Allen 1983
Pachymetopon blochi	Hottentot	larvae	0.42	24 h	Brownell 1980
Pimephales promelas	Fathead minnow		0.75-3.4	96 h	Thurston et al. 1983
Poecilia reticulata	Guppy	fry	0.66-2.62	24 h	Rubin and Elmaraghy 1977
		fry	1.06-1.38	48 h	Rubin and Elmaraghy 1977
		fry	1.00-1.26	96 h	Rubin and Elmaraghy 1977
Salmo clarki	Cuthroat trout		0.5-0.8	96 h	Thurston et al. 1978
		adults	0.30-0.33	96 h	Thurston et al. 1981
S. salar	Atlantic salmon		0.04-0.23	24 h	Alabaster et al. 1979
		2-year smolt	0.80-2.0	24 h	Alabaster et al. 1983

Table 5.1. Acute effects of un-ionized ammonia on aquatic animals (concluded).

Organism		Life Stage	Conc. (mg NH_3/L)	Exposure period	Reference
Sciaenops ocellatus	Red drum	post hatch larvae	0.39	96 h	Holt and Arnold 1983
Stizostedion vitreum	Walleye		0.66	96 h	Arthur et al. 1987
Tilapia aurea	Tilapia		2.4	48 h	Redner and Stickney 1979
Invertebrates					
Asellus racovitzai	Isopod		5.02	96 h	Arthur et al. 1987
Callibaetis skokianus	Mayfly		3.90	96 h	Arthur et al. 1987
Cherax quadricarinatus	Australian crayfish	juvenile	2.02	24 h	Meade and Watts 1995
			1.63	48 h	Meade and Watts 1995
			0.98	98 h	Meade and Watts 1995
Crangonyx pseudogracilis	Amphipod		3.12	96 h	Arthur et al. 1987
Crassostrea virginica	American oyster		8.36-23.46	96 h	Epifanio and Srna 1975
Helisoma trivolvis	Snail		2.37	96 h	Arthur et al. 1987
Macrobrachium rosengergii	Freshwater prawn	3-8 days old	0.66-3.58	24 h	Armstrong et al. 1978
			0.26-1.35	144 h	Armstrong et al. 1978
Mercenaria mercenaria	Quahog clam		3.06-4.49	96 h	Epifanio and Srna 1975
Musculium transversum	Fingernail clam		1.10	96 h	Arthur et al. 1987
Orconectes nais	Crayfish		3.2	24 h	Hazel et al. 1979
O. immunis	Crayfish		18.3	96 h	Arthur et al. 1987
Perna viridis	Green mussel		6.2	48 h	Reddy and Menon 1979
			11.6	96 h	Reddy and Menon 1979
Physa gyrina	Snail		1.95	96 h	Arthur et al. 1987
Philartcus quaeris	Caddisfly		10.1	96 h	Arthur et al. 1987

In conclusion, aquatic plants appear to be relatively tolerant of ammonia as compared to fish and invertebrates. Nitrates are relatively non-toxic and, therefore, their toxicity to aquatic plants has not been tested. However, with the growing interest in the use of aquatic macrophytes to treat wastewater (Pries 1994; Kadlec and Knight 1996), further studies are needed to identify ammonia (as well as nitrate and nitrite) sensitivity of fresh and salt water plant species.

Aquatic animals

The scientific literature on ammonia toxicity to aquatic animals is extensive. A complete review of that large volume of literature is beyond the scope of this report and only a small selection of the literature is presented.

In most surface waters, total ammonia concentrations greater than about 2 mg/L are toxic to aquatic animals (Mueller and Helsel 1996), although this varies among species and life stages (Table 5.1). Because seawater is typically more alkaline than freshwater and thus has a greater proportion of total ammonia in the un-ionized form, total ammonia concentrations eliciting a toxic response are likely higher in seawater than in freshwater (Environment Canada 1997b). With respect to the toxic un-ionized form, mean 96-hour LC_{50} values for fishes range from 0.083 to 3.8 mg NH_3/L. The 48-hour LC_{50} values range from 0.51 to 2.4 mg NH_3/L and 24-hour LC_{50} values range from 0.04 to 2.62 mg NH_3/L (Table 5.1). The cause of death appears to be an increase in permeability of the fish to water such that the water intake ultimately exceeds the maximum sustained urine production (NRC

Table 5.2. Acute and chronic effects of nitrate on aquatic animals. All endpoints were LC_{50} (the concentration of NO_3^- at which 50% of the test organisms died) unless otherwise noted.

Organism		Life Stage	Conc. (mg NO_3/L)	Exposure period	Reference
Fish					
Centropristis striata	Gulf Black sea bass		10629	96 h	Pierce et al. 1993
Ictalarus punctatus	Channel catfish		6200	96 h	Colt and Tchobanoglous 1976
Lepomis macrochirus	Bluegill		1860-8860	96 h	Trama 1954
Micropterus treculi	Guadelupe bass	fingerlings	1261	96 h	Tomasso and Carmichael 1986
Monacanthus hispidus	Planehead filefish		2538	96 h	Pierce et al. 1993
Oncorhynchus mykiss	Rainbow trout	fingerlings	6000	96 h	Westin 1974
		eggs	10	--	Kincheloe et al. 1979
		eggs	5	--	Kincheloe et al. 1979
O. tshawytscha	Chinook salmon	fry	5800	96 h	Westin 1974
		fry	20	--	Kincheloe et al. 1979
Poecilia reticulata	Guppy		797-886	96 h	Rubin and Elmaraghy 1977
Pomacentrus leucostritus	Beaugregory		>13286	96 h	Pierce et al. 1993
Raja eglanteria	Clearnose skate		>4252	96 h	Pierce et al. 1993
Trachinotus carolinus	Florida pompano		4429	96 h	Pierce et al. 1993
Amphibians					
Bufo americanus	American toad	tadpoles	60-175	96 h	Hecnar 1995
B. bufo	Common toad	tadpoles	1704	96 h	Xu and Oldham 1997
			1637	168 h	Xu and Oldham 1997
Litoria caerulea	Tree frog	tadpoles	100[1]	16 d	Baker and Waights 1994
		tadpoles	40[2]	16 d	Baker and Waights 1994
Pseudacris t. triseriata	Western chorus frog	tadpoles	76	96 h	Hecnar 1995
		tadpoles	11[3]	24 h	Hecnar 1995
		tadpoles	44[4]	100 d	Hecnar 1995
Rana pipiens	Northern leopard frog	tadpoles	100	96 h	Hecnar 1995
		tadpoles	44[4]	100 d	Hecnar 1995
R. clamitans	Green frog	tadpoles	144	96 h	Hecnar 1995
Triturus vulgaris	Smooth newt	larvae	77.5-387.5[5]	24 h, 48 h or 72 h	Watt and Oldham 1995

[1] Endpoint is death
[2] Endpoint is an Effective Concentration (EC) (behaviour change)
[3] Endpoint is an EC (physical and behavioural changes)
[4] Endpoint is an EC (survival during metamorphosis)
[5] Endpoint is an EC (size of metamorphs, feeding rate)

1979; Environment Canada 1984). Sublethal exposure to ammonia has been reported to cause adverse physiological and histopathological effects in fish (NRC 1979). Less information is available on the toxic effects of ammonia to invertebrates and benthic organisms. Mean 48 and 96-hour LC_{50} values for freshwater invertebrates range from < 1 to 23.46 mg NH_3/L (Hickey and Vickers 1993; Table 5.1).

Nitrate is considerably less toxic to aquatic organisms than ammonia or nitrite and, as a result, has been less studied (Russo 1985; Table 5.2). For adult fish, LC_{50} values range from 800 to 12 000 mg NO_3^-/L (Colt and Tchobanoglous 1976; Tomasso and Carmichael 1986). However, nitrate concentrations in the range of 1 to 10 mg NO_3^-/L are lethal to eggs and, to a lesser extent, fry of salmonid fish (Kincheloe et al. 1979). In addition, Kincheloe et al. (1979) showed that 31% of rainbow trout (*Oncorhynchus mykiss*) eggs and 15% of fry died when exposed to 2.3 mg NO_3^-/L. Nitrate exposure in tadpoles from various amphibian species has led to lethality at concentrations as low as 60 mg/L (Hecnar 1995; Table 5.2). However, the behaviour and survivorship of some species were affected by concentrations as low as 11 mg NO_3^-/L (Hecnar 1995). Furthermore, tadpoles exposed to 11-44 mg NO_3^-/L for 24-h had reduced feeding activity and weight loss, swam less vigorously, showed a delayed response to prodding, and displayed disequilibrium and eventually paralysis (Hecnar 1995). Signs of developmental abnormalities were also observed and included: development of bulges in the head and thorax regions; eye deformities; complete loss of pigment; subdermal edema resulting in a large globular body shape; digestive-tract deformities; and bent tails (Hecnar 1995). In addition, the size at metamorphosis decreased with increasing nitrate in the smooth newt (*Triturus vulgaris*) (Watt and Oldham 1995), which could ultimately affect the population dynamics of these newts. It is interesting to note that amphibians appear to be more sensitive to nitrate exposure than fish (Table 5.2). Nitrate toxicity experiments conducted on invertebrates such as two species of caddisfly larvae showed relatively high LC_{50} values of 504.4 mg NO_3^-/L for *Cheumatopsyche pettiti* and 432.4 mg NO_3^-/L for *Hydropsyche occidentalis* (Camargo and Ward 1992).

Under some conditions, nitrate and ammonia can be transformed to nitrite (Marco and Blaustein 1999). Nitrite is highly toxic, more so than nitrate. It is suspected that nitrite can reach lethal levels in aquatic culture systems, ponds, waters receiving municipal wastewater effluents, and any natural system where animal biomass is high (Huey and Beitinger 1980b). However, because it rapidly oxidizes to nitrate and is therefore rarely present in high concentrations naturally, its toxicity has not been well studied. Of the few studies that exist, the 96-hour LC_{50} values for fish (0.19 to 188 mg NO_2^-/L; Table 5.3) are, on average, higher than those reported for ammonia. Moreover, of the fish species studied, rainbow trout (*O. mykiss*) appears to be the most sensitive. Huey and Beitinger (1980b) exposed larvae of salamanders (*Ambrystoma tesnaum*) to nitrite and determined an LC_{50} of 1.09 mg NO_2^-/L. Sublethal concentrations of nitrite (11.5 mg NO_2^-/L) induced behavioural and morphological changes, retarded development, and altered the age at emergence in the cascades frog (*Rana cascadae*) (Marco and Blaustein 1999). In addition, nitrites transform haemoglobin into methaemoglobin, decreasing the oxygen carrying capacity of the blood: nitrite concentrations as low as 1 mg/L have been shown to increase the amount of methaemoglobin in the blood of bullfrog (*Rana catesbiana*) tadpoles (Huey and Beitinger 1980a). Other behavioural modifications (occupying shallow waters, bobbing behaviour, and air gulping) have been noticed in amphibians in response to high nitrite concentrations (11 – 50 mg NO_2^-/L) in surface waters (Huey and Beitinger 1980a; Marco and Blaustein 1999).

Terrestrial Plants

Anthropogenic inputs of N to vegetation are associated with air pollution, particularly nitrogen oxides, un-ionized ammonia, ammonium and nitrate. Of these compounds, nitrogen dioxide, un-ionized ammonia and ammonium are all highly phytotoxic. Nitrogen-containing air pollutants can affect vegetation indirectly via chemical reactions in the atmosphere immediately surrounding the plant, or directly from uptake after deposition on vegetation, soil or water surfaces (WHO 1997a).

Nitrogen toxicity exhibits a variety of symptoms including abscission of foliage and reductions in total assimilatory capacity (Miller et al. 1979 cited in Jacobson et al. 1990); formation of larger, thin walled cells and a "succulent" type of growth (Roelofs et al. 1985; Teshow and Anderson 1989); needle yellowing and lesions (Van Dijk and Roelofs 1988); reduction in drought hardiness due to loss of control of transpiration by stomata and to an increase in shoot/root ratios (Lucas 1990; Atkinson et al.

Table 5.3. Acute and chronic effects of nitrite on aquatic animals. All endpoints are LC_{50} (the concentration of NO_2^- at which 50% of the test organisms died) unless otherwise indicated.

Organism		Life Stage	Conc. (mg NO_2/L)	Exposure period	Reference
Fish					
Acipenser brevirostrum	Shortnose sturgeon	fingerlings	11.3	96 h	Fontenot et al. 1998
Anguilla anguilla	European eel		143.7	96 h	Kamstra et al. 1996
Ictalurus punctatus	Channel catfish		7	96 h	Palachek and Tomasso 1984
Micropterus treculi	Guadelupe bass	fingerling	187.6	96 h	Tomasso and Carmichael 1986
M. salmoides	Largemouth bass		140	96 h	Palachek and Tomasso 1984
Oncorhynchus mykiss	Rainbow trout	fingerling	0.06[1]	6 mon.	Wedemeyer and Yasatake 1978
		fingerling	0.015[2]	6 mon.	Wedemeyer and Yasatake 1978
		fingerling	0.06[1]	10 d	Russo et al. 1974
		fingerling	0.19	96 h	Russo et al. 1974
O. tshawytscha	Chinook salmon	fry	2.9	96 h	Westin 1974
Amphibians					
Rana cascadae	Cascades frog	tadpoles	11.5[3]	14 d	Marco and Blaustein 1999
Rana catesbiana	Bullfrog	tadpoles	1-50[4]	24 h	Huey and Beitinger 1980a
Tilapia aurea	Tilapia		16	96 h	Palachek and Tomasso 1984
Ambrystoma tesnaum	Smallmouth salamander	larvae	6.6	96 h	Huey and Beitinger 1980b
		larvae	1.09	96 h	Huey and Beitinger 1980b
Invertebrates					
Cherax quadricainatus	Australian crayfish	juveniles	42.9	24 h	Meade and Watts 1995
		juveniles	37.1	48 h	Meade and Watts 1995
		juveniles	25.9	96 h	Meade and Watts 1995

[1] Endpoint is a NOEC (No observed effect concentration)
[2] Endpoint is an Effective Concentration (EC) (methaemoglobin changes)
[3] Endpoint is an EC (developmental and behavioural changes)
[4] Endpoint is an EC (% methaemoglobin)

1991; Fangmeijer et al. 1994); and reduction in frost hardiness due to the disruption of the production of anti-freeze proteins and the utilization of carbohydrates (Cape et al. 1991; Hall et al. 1997). The adverse effects of N-containing compounds may be due to their interference with cellular acid-base regulation as they can influence cellular pH both before and after assimilation (Raven 1988; Wollenweber and Raven 1993).

Nitrate is thought to be essentially non-toxic to plants (Teshow and Anderson 1989; Yaremcio 1991; WHO 1997a). Certain plant species can accumulate nitrate to concentrations that present health complications in their consumers (see Terrestrial Animal section). Nitrate accumulation in plants results from uptake in excess of their capacity to reduce and assimilate nitrate into amino acids (Environment Canada 1997b). Acid precipitation, which is caused by sulfate and nitrate ions, can injure vegetation through the precipitation of major ions, increased water loss and decreased photosynthesis. However, the effects of acid precipitation do not appear to be due to nitrogen (or sulfate) directly but to the decrease in foliar pH.

When in excess, nitrogen oxides may cause toxicity (Table 5.4). The greatest direct effect on vegetation lies in the interaction of NO_2 with other pollutants and in disrupting balances with other nutrients (WHO 1997a). Woody plant species that lack the necessary enzymes to metabolize the nitrate and nitrite formed in the leaves by the absorption of nitrogen oxides may be particularly

Table 5.4. Acute and chronic effects of nitrogen (ammonia, nitrogen dioxide, and nitric oxide) on terrestrial plants.

Organism		Effect	Conc. ($\mu g/m^3$)	Exposure period	Reference
Ammonia					
Amaranthus retroflexus	Pigweed	Slight injury	8.4	4 h	NRC 1979
Brassica alba	Mustard weed	Severe injury	2.1	4 h	NRC 1979
B. rapa	Turnip	Leaf injury	2000	2 h	Benedict and Breen 1955
Coleus	Coleus	Leaf lost colour	28.2	1 h	NRC 1979
Fagopyrum esculentum	Buckwheat	50% foliar injury	704	5 min	NRC 1979
Helianthus	Sunflower	Injury	28.2	1 h	NRC 1979
Lepidium sativum	Garden cress	Reduced biomass	1000	2 wk	Van Haut and Prinz 1979
Lolium perenne	Perennial rye grass	Impact acid/base regulation	213	5 d	Wollenweber and Raven 1993
Lycoperiscon esculentum	Tomato	Injury	28.2	1 h	NRC 1979
		50% foliar necrosis	176	4 min	NRC 1979
Malus sylvestris	Apples	Dark discolouration around lenticels	>282	–	NRC 1979
Nicotiana tabacum	Tobacco	50 % foliar injury	704	8 min	NRC 1979
Pinus sylvestris	Scotch pine	Increased loss of water	53	9 mnth	Dueck et al. 1990
		Increased glutamine synthetase activity	60	14 wk	Pérez-Soba et al. 1990
Prunus persica	Peach fruit	Darkening of skin	140.8	–	NRC 1979
Pseudotsuga menziesii	Douglas fir	Imbalanced nutrient status	180	13 wk	Van der Eerden et al. 1991
Raphanus sativus	Radish seeds	Mortality	704	16 h	NRC 1979
Secale cereale	Spring rye seeds	52% germination decrease	176	16 h	NRC 1979
		Mortality	704	4 h	NRC 1979
Violion caninea	Violet	Imbalanced nutrient status	50	8 mnth	Dueck and Elderson 1992
Nitrogen dioxide					
Citrus sinesis	Orange	Increased fruit drop	120-500	9.5 mnth	Thompson et al. 1970
Glycine max	Soya bean	Enhanced dark respiration and photosynthesis	375	35 h	Sabarathnam et al. 1988
Helianthus annus	Sunflower	Reduced net assimilation rate	375	2 wk	Okano et al. 1985
Hordeum vulgare	Barley	Growth stimulation	10-43	130 d	Adaros et al. 1991
Lolium perenne	Perennial rye grass	Lower glutamate dehydrogenase activity	125	140 d	Wellburn et al. 1981
Lycoperiscon esculentum	Tomato	Lower leaf nitrate concentration	940	19 d	Taylor and Eaton 1966
Medicago sativa	Alfalfa	Inhibition of photosynthesis	1100	1.5 h	Hill and Bennet 1970
Phaseolus vulgaris	Common bean	Increased dark respiration	120	22 d	Sandhu and Gupta 1989
Picea rubens	Red spruce	Increased nitrate reductase activity	140	1 d	Norby et al. 1989
Pinus strobus	Eastern white pine	Slight needle necrosis	100	5 d	Yang et al. 1983
Pinus sylvestris	Scotch pine	Increased nitrate reductase activity	160	7 d	Wingsle et al. 1987
Pisum sativum	Pea	Emission of stress ethylene	850	7 h	Mehlhorn and Wellburn 1987
Polytrichum formosum	Polytrichium moss	Injury and changes in growth	122	37 wk	Bell et al. 1992

Table 5.4. Acute effects of nitrogen on terrestrial plants (concluded).

Organism		Effect	Conc. ($\mu g/m^3$)	Exposure period	Reference
Nitric oxide					
Avena sativa	Oats	Reduced photosynthesis	750	1 h	Hill and Bennet 1970
Capsicum annum	Sweet pepper	Decreased nitrate reductase	1875	18 h	Murray and Wellburn 1980
Ctendium molluscum	Bryophyte	Increased dark respiration	44	8-24 h	Morgan et al. 1992
Lactuca sativa	Lettuce	Induction of nitrate reductase	375	8 d	Besford and Hand 1989
Lactuca sativa		Reduced photosynthesis	2500	10 min	Capron 1989
Lactuca sativa		Reduced biomass	635	16 d	Capron et al. 1991
Lycoperiscon esculentum	Tomato	Induction of nitrate reductase	500	28 d	Wellburn et al. 1980
		Reduced biomass	500	5 d	Bruggink et al. 1988
Medicago sativa	Alfalfa	Reduced photosynthesis	750	1 h	Hill and Bennet 1970
Pisum sativum	Pea	Increased ethylene release	188	7 h	Mehlhorn and Wellburn 1987

susceptible to injury (Amundson and McLean 1982). Nitrogen dioxide in concentrations of 100-140 µg NO_2/m^3 caused slight needle necrosis in eastern white pine (Yang et al. 1983), increased dark respiration in beans (Sandhu and Gupta 1989), and increased nitrate reductase activity in red spruce (Norby et al. 1989). Nitrogen dioxide at a concentration of 375 µg NO_2/m^3 increased dark respiration and photosynthesis in soya beans (Sabarathnam et al. 1988) and reduced net N assimilation rate in sunflowers (Okano et al. 1985). NO_2 concentrations as high as 900-1100 µg NO_2/m^3 decreased leaf nitrate concentrations in tomato plants (Taylor and Eaton 1966) and inhibited photosynthesis in alfalfa (Hill and Bennet 1970).

Terrestrial plants may also show a toxic response to nitric oxide (Table 5.4). Leaves are more sensitive to nitric oxide injury under low light or dark conditions (Taylor 1973), as the process that reduces nitrates operates more slowly when plants are not photosynthesizing (Ramade 1987). Nitric oxide has been shown to increase dark respiration at concentrations as low as 44 µg NO/m^3 in bryophytes (Morgan et al. 1992), reduce plant mass in tomatoes at concentrations of 500 µg NO/m^3 (Bruggink et al. 1988), and inhibit photosynthesis in lettuce at concentrations of 2 500 µg NO/m^3 (Capron 1989).

Ammonium (NH_4^+) in plant tissues is known to be toxic at quite low concentrations. Ammonium toxicity is generally characterized by an immediate reduction in growth rate, wilting, marginal necrosis, inter-veinal chlorosis of terminal leaves and, ultimately, death of the entire plant (Maynard and Barker 1969). For example, exposure of the roots of tomato seedlings to an ammonium solution as the sole N source resulted in reduced growth, a bruised root system and considerable wilting of the leaves, which developed marginal necrosis (Pierpont and Minotti 1977). A significant effect of increased atmospheric ammonium is that it may stimulate leaching; in this regard, ammonium has been shown to increase leaching of potassium from hardwoods (Roelofs et al. 1985; Leonardi and Flückiger 1989) and calcium and magnesium ions from conifers (Wilson 1992). This leaching, in turn, can lead to nutrient imbalances as shown in Douglas fir at concentrations of 180 µg NH_4^+/m^3 (Van der Eerden et al. 1991) and violets at concentrations of 50 µg NH_4^+/m^3 (Dueck and Elderson 1992). Atmospheric ammonium ions may also contribute to a recent decline of forests by affecting frost hardiness. Changes in frost hardiness of only a few degrees Celsius could significantly increase the risk of frost damage (Friedland et al. 1984; Sheppard et al. 1989; Cape et al. 1991; Hall et al. 1997).

Foliar uptake of un-ionized ammonia (NH_3) is also possible and high ambient concentrations can be toxic. Ammonia may exert direct effects on plant structure and function and indirect effects on soil conditions (Environment Canada 1997b; Table 5.4). Ammonia is an N source for the synthesis of proteins and, although the use of ammonia in the synthesis of organic molecules can be regarded as a process for storing a valuable nutrient, it is also important for detoxifying a potentially toxic chemical, namely ammonia. Excessive uptake of ammonia can therefore stress carbohydrate metabolism. Ammonia is also an inhibitor of photosynthetic phosphorylation, which results in reduced carbohydrate production and therefore, reduced growth (Teshow and Anderson 1989). The toxic effects of ammonia are most often seen in older leaves or needles. Symptoms often resemble those induced by drought, salts, some plant diseases, or other pollutants. Symptoms specific to ammonia include dark spots or the complete blackening of leaves, rusty marginal spotting, and dark brown lesions between the veins on leaves, which turn black the next day followed by the entire leaf drying up (NRC 1979). Colour changes in fruit and vegetable skins and brownish to black lesions also occur (NRC 1979). Injury symptoms on broad-leaved woody plants exposed to high concentrations of ammonia usually begin as large, dark green, water-soaked areas that darken after several hours to brownish-grey or black necrotic lesions widely scattered over the leaf surface (NRC 1979). On trees or shrubs with crowded or overlapping leaves, injury may be confined to particular sections of the leaf. In addition, foliar lesions can occasionally turn orange, purple, or reddish-brown, mimicking fall colouration. Conifer foliage exposed to ammonia darkens to shades of grey-brown, purple, or black and the entire part of the needle exposed to the gas is usually affected (Dueck et al. 1990). Symptoms of injury are more variable on herbaceous plants than on woody species, ranging from irregular, bleached, bifacal, necrotic lesions to reddish inter-veinal streaking or dark upper-surface discolouration (Van der Eerden 1982; Environment Canada 1984; Teshow and Anderson 1989; Hall et al. 1997; WHO 1997a; Table 5.4). In addition, concentrations as high as 704 µg NH_3/m^3 applied to radish and rye seeds caused mortality (NRC 1979). As with ammonium, a major effect of ammonia exposure to plants is increased sensitivity to cold and frost (Cape et al. 1991). Thus, elevated atmospheric ammonia concentrations may also lead to forest decline as a result of lipid saturation by ammonia in cell membranes, thereby increasing their permeability and decreasing their flexibility (Van der Eerden 1982).

Although P is not directly toxic in the forms and amounts found naturally in the environment (Switzer-Howse and Coote 1984), plants supplied with a small excess of P often develop purple to rusty-brown lesions on their dark green lower leaves (Jones 1998). Growth is not impaired when excess P is supplied in small amounts. However, plants supplied with a large excess of P lack vigour and have short stout stems. The leaves are dark green in colour and the blades of flower leaves may be almost covered by reddish-purple lesions. Eventually, the tips and margins of the blades turn brown, wither, curl upwards and die. Clark (1982) and Furlani et al. (1986a, b) described reddish purple lesions on sorghum plants growing in solution cultures, potted sand or soil, or in the field and referred to them as "red-speckling". The concentration of P that induced this response depended on the cultural method, cultivar, age of the plants, and source of phosphate. With a susceptible cultivar, "red-speckling" appeared 3 days after 2.18 µg/L potassium dihydrogen phosphate was added to solution cultures of 7-day-old plants (Furlani et al. 1986b) or after 0.58 µg/L ammonium dihydrogen phosphate was added to

Table 5.5. Acute and chronic effects of nitrogen (nitrate and ammonia) on terrestrial organisms.

Organism	Endpoint	Conc. (mg/m^3)	Exposure period	Reference
Nitrate				
Guinea pigs	Lung structure. No significant effects	1	4 wk	Busch et al. 1986
Rat	Lung structure. No significant effects	1	4 wk	Busch et al. 1986
Ammonia				
Bats	No adverse physiological effects	59.8-1302.4	--	NRC 1979
Birds (Starling; Sparrow; Pigeon)	LC_{50}	1600	7 min	NRC 1979
Cat	Severe dysponea, anorexia, dehydration, bronchopneumonia after 7 days	700	10 min	Dodd and Gross 1980
Cattle (dairy)	High morbidity and mortality rates	182-1400	--	NRC 1979
Mouse	30% mice dead; nasal, eye and breathing problems	3185	1 h	MacEwen and Vernot 1972
Pigs	Excessive nasal, lacrimal and mouth secretions	70.4-105.6	5 wk	NRC 1979
Poultry	Reduced food consumption and weight gain; air sacculitis; increased susceptibility to respiratory diseases; general discomfort	14.1-35.2	--	NRC 1979
Poultry	Reduced egg production, tracheitis, keratoconjunctivitis (ocular disorder)	42.2-70.4	--	NRC 1979
Rabbit	No signs of toxicity; no lung abnormalities	40	24 hr/day for 114 days	Coon et al. 1970
Swine	Reduced growth rate	>70.4	--	NRC 1979
Swine	Frothing of the mouth, excessive secretion, and irregular breathing	196	36 h	Stombaugh et al. 1969

solution cultures of 2-day-old seedlings of *Sorghum bicolor* (Grundon et al. 1987). The "red-speckling" of P toxicity can be confused with the reddish purple or brown lesions developed by plants suffering from a severe deficiency of potassium or magnesium. However, P toxicity lesions are irregular in outline and the brown necrosis is confined to the margins.

Terrestrial Animals

Nitrogen toxicity in terrestrial animals is associated either with air pollution (nitrogen dioxide, ammonia or nitrate) or the consumption of vegetation with high nitrate levels. However, water-borne nitrates and nitrites have caused the induction of methaemoglobinaemia in young animals (WHO 1977).

With respect to nitrogen toxicity from atmospheric sources, studies of nitrogen dioxide have largely described effects on the respiratory tract (Cavanagh and Morris 1987; WHO 1997a), but extrapulmonary effects have also been briefly studied (e.g., WHO 1997a). Nitrogen dioxide exposure causes structural alterations in lungs, including emphysema. Ammonia studies have mostly focused on inhalation and irritation effects (Environment Canada 1984; Environment Canada 1997b). Table 5.5 shows the effects of nitrate and ammonia gas inhalation on mammals. There are no reported effects of ammonium aerosol inhalation on various mammals (Rothenberg et al. 1986; Schlesinger et al. 1990). Nitrate may act by altering intracellular pH, which can cause serious health effects (WHO 1997a). Only one inhalation study has been reported for mammals (Busch et al. 1986) and there is no information on the effect of nitrate on reptiles (Rouse et al. 1999) (Table 5.5).

Table 5.6. Plants used as livestock feed that accumulate nitrate. (Data from Bradley et al. 1940; Osweiler et al. 1985; Mulligan and Munro 1990; Yaremcio 1991; Wohlgemuth and Casper 1992; Steppuhn et al. 1994)

CROPS	WEEDS
Alfalfa (*Medicago sativa*)	Canada thistle (*Cirsium arvense*)
Barley (*Hodeum vulgare*)	Dock (*Rumex venosus*)
Canola (*Brassica rapa*)	Fireweed (*Epilobium angustifolium*)
Corn (*Zea mays*)	Jimsonweed (*Datura stramonium*)
Flax (*Linum usitatissimum*)	Johnson grass (*Sorghum halepense*)
Millet (*Panicum miliaceum*)	Kochia (*Kochia scparia*)
Oats (*Avena sativa*)	Lambsquarter (*Chenopodium album*)
Rape (*Brassica napus*)	Nightshade (*Solanum* sp.)
Rye (*Secale cereale*)	Pigweed (*Amaranthus* sp.)
Soyabean (*Glycine max*)	Russian thistle (*Salsola kali*)
Sorghum (*Sorghum* sp.)	Smartweed (*Polygonum acre*)
Sudangrass (*Sorghum sudanense*)	Wild sunflower (*Helianthus* sp.)
Sugar beets (*Beta vulgaris*)	
Sweet clover (*Melilotus alba*)	
Wheat (*Triticum aestivum*)	

Nitrate poisoning as a result of plant ingestion occurs when animals consume in excess of 0.35 to 0.45% nitrate in their diet (Kvasnicka and Krysl 1996). Livestock fed on feed containing 0 to 0.5 % nitrate (dry weight) show no signs of toxicity. However, livestock given feed with nitrate levels of 0.5 to 1.0 % (dry weight) may show sublethal symptoms. Nitrate concentrations >1.0% (dry weight) have caused death of livestock (Neilson 1974; Osweiler et al. 1985; Yaremcio 1991; Wohlgemuth and Casper 1992).

Nitrates can accumulate to high levels in certain plant species, in particular common plants used for livestock feed (Table 5.6). In general, annual forage crops tend to accumulate greater amounts of nitrate than perennial forages. A variety of conditions are associated with nitrate-enrichment of vegetation (e.g., drought, insect damage, hail, cloudy weather, crowding, shading, application of nitrogen-containing fertilizers, and farm management) (Bradley et al. 1940; Yaremcio 1991; Steppuhn et al. 1994; Kvasnicka and Krysl 1996; Lester 1998). The frequent use of nitrogen-fertilizers in recent years has resulted in increased incidence of nitrate poisoning (Kvasnicka and Krysl 1996). Plants may also accumulate livestock-toxic levels of nitrate (up to 4.7 % dry weight NO_3) if rooted on soils heavily contaminated by animal excreta, for example around water tanks, ponds and salt licks and along frequently used trails (Williams 1989).

Nitrate in vegetation is not toxic to livestock until it is converted to nitrite in the animal's digestive tract. Nitrite is absorbed into the blood and converts haemoglobin to methaemoglobin, which has no oxygen carrying capacity (Yaremcio 1991). Cattle and sheep are more susceptible to poisoning because micro-organisms in their rumen favour this conversion to nitrite (Bradley et al. 1940; Osweiler et al. 1985). Horses, pigs, and poultry are less susceptible to nitrate poisoning because they convert nitrate to nitrite in the intestine, where there is less opportunity for nitrites to be absorbed by the blood (Neilson 1974; Yaremcio 1991).

Acute nitrate poisoning usually occurs within a half to four hours after consuming toxic levels (> 1.0 % dry weight) (Wohlgemuth and Casper 1992). Signs of acute poisoning in livestock are increased heart rate; quickened respiration; muscle tremors; vomiting; weakness; blue-grey mucous membranes; blindness; excess saliva and tear production; depression; laboured or violent breathing; staggered gait; frequent urination; low body temperature; disorientation; and an inability to get up (Emerick 1974; Cowley and Collings 1977; Yaremcio 1991; El-Bahri et al. 1997). Animals are often found lying down after only a short struggle. In most cases of acute poisoning, animals are found dead before any signs of toxicity are observed.

Sublethal responses to nitrate affect livestock production and are not often recognized. They include lower milk production, reduction in weight gain and lower resistance to disease, resulting in poor animal performance and a general lack of health (Emerick 1974; Hibbs et al. 1978; El-Bahri et al. 1997; Lester 1998). Sublethal doses of nitrate also increase the incidence of stillborn calves, abortions, retained placentas, and cystic ovaries (Hibbs et al. 1978; Johnson et al. 1983; Putnam 1989). In addition, nitrate blocks the conversion of beta-carotene to true vitamin A and alpha-tocopherol to vitamin E, resulting in photosensitivity and other chronic disorders (Lester 1998).

Nitrate poisoning is treatable with intravenous administration of methylene blue shortly after ingestion of nitrate-enriched plants (Osweiler et al. 1985; Yaremcio 1991). Careful use of nitrogen-fertilizers, an awareness of which plants accumulate nitrate, silage moisture levels, the effect of weather, and cattle feed management can help reduce losses from nitrate poisoning. Prevention of nitrate poisoning is best achieved by controlling the type and quantity of forage offered to livestock. Most feeds that contain nitrate can be fed to cattle if managed properly.

Water-borne nitrate or nitrite can also cause problems for terrestrial animals. Methaemoglobinaemia is the most studied problem, although nitrate has induced prenatal mortality, resorptions, and decreased birthweight in rat and guinea pig pups whose mothers received drinking water containing sodium nitrite (Alexandrov and Jänisch 1971; Shuval and Gruener 1972; WHO 1977). Nitrate induced increases in methaemoglobin levels in rabbits have been found to increase cardiac activity (Garbuz 1971). In adult rats, elevated sodium nitrite levels in drinking water increased methaemoglobin levels but not to the point of producing overt toxic effects (Druckrey et al. 1963; Behroozi et al. 1971).

Human Health

The effect of controlled and accidental acute exposure to ammonia on human health has been widely studied (see WHO 1986 for a complete review). The data show serious health effects of ammonia; however, further research is required to identify exposure time and concentration guidelines for spill situations and to identify long-term health problems associated with acute toxic exposure. Human contact with ammonia may be in the form of gas, anhydrous liquid, or aqueous solution (Environment Canada 1997b). Inhalation is considered to be the primary route for human exposure and effects can range from mild irritation to severe corrosion of sensitive membranes of the eyes, nose, throat and lungs. The most dangerous consequence of exposure to the gas is pulmonary edema followed by severe irritation to moist skin surfaces (WHO 1986; Environment Canada 1997b; WHO 1997a;).

In addition to ammonia, nitrate can pose a serious problem to human health. In Canada, a maximum acceptable concentration of 45 mg/L for nitrate [10 mg NO_3^--N /L] in drinking water is recommended (Health Canada 1996). Nitrite, which is directly toxic and is produced from nitrate, has a recommended maximum acceptable concentration of 3.2 mg NO_2^-/L [0.97 mg NO_2^--N/L] in drinking water (Health Canada 1996). Nitrate levels in Canadian municipal water supplies are generally less than

5 mg NO_3^-/L (Health Canada 1996). Concentrations in ground water supplies vary across the country (Table 5.10).

Ingestion of high quantities of nitrate or nitrite may result in methaemoglobinaemia ("Blue Baby Syndrome"), a condition resulting from the oxidation (by nitrite) of ferrous (Fe^{2+}) to ferric (Fe^{3+}) iron in haemoglobin, the oxygen carrier of mammalian blood. The resulting methaemoglobin has no oxygen carrying capacity. Symptoms are cyanosis, asphyxia, and death (Health Canada 1992; WHO 1997a). The most sensitive subpopulation is infants less than three months of age. Reasons for the greater susceptibility of young infants include their readily oxidizable fetal haemoglobin and their depressed methaemoglobin capacity to secrete gastric acid and increased susceptibility to gastro-enteritis, both of which permit nitrate-reducing bacteria in the gastrointestinal tract to convert nitrate to nitrite (Craun et al. 1981).

For bottle-fed infants, water used in the preparation of infant formula is generally the main source of nitrates (Health Canada 1996). Consumption of 0.6 L of drinking water containing 4.5 mg/L nitrate could contribute 2.7 mg NO_3^-/d or about 0.7 mg NO_3^-/kg body weight for a 3 to 5 kg infant (Health Canada 1992). For adults, food is generally the main source unless drinking water concentrations are elevated. Vegetables such as beets, celery, lettuce, radishes, and spinach contribute significantly to an adult's dietary intake of nitrate (Health Canada 1992), with nitrate levels ranging from 1 700 to 2 400 mg NO_3^-/kg food (Corre and Breimer 1979). Spinach can easily accumulate nitrates from the soil and consumption of spinach grown in soils heavily fertilized with nitrate leads to human health risks (methaemoglobinaemia) (Ramade 1987). There have been cases in Europe of methaemoglobinaemia due to consumption of spinach contaminated with excess nitrates (Ramade 1987). Small amounts of nitrates may also be present in fish and dairy products such as cheese, baked goods, and cereals. The total daily intake of nitrates by adults is estimated to be 51 mg (44.3 mg from food and 6.8 mg from drinking water containing nitrate at a concentration of 4.5 mg/L), or 0.7 mg/kg body weight for a 70-kg adult (Health Canada 1992).

Ingested nitrate is readily absorbed in the upper small intestine and is distributed rapidly throughout the body of which roughly 25% is recirculated into the saliva. Once in the saliva, 20% of the recirculated nitrate is reduced to nitrite by the oral microflora (Health Canada 1992). Nitrite production increases with age (Turek et al. 1980) and is enhanced during bacterial infections causing diarrhoea (Hartman 1983). Nitrites are rapidly absorbed both in the stomach and in the small intestine. In the human stomach, nitrites can react with nitro-satiable compounds, such as amides and amines, to form N-nitroso compounds (Broitman et al. 1981). Some N-nitroso compounds are potent carcinogens and corrosive substances in animal species (Broitman et al. 1981; Ramade 1987), and are therefore probably carcinogenic in humans (Fraser 1985). Some evidence for an association between the ingestion of nitrate and the incidence of stomach cancer has been obtained in descriptive epidemiological studies (Hill et al. 1973; Fraser and Chilvers 1981; Clough 1983; Gilli et al. 1984). Yet, other studies indicate no association between the ingestion of nitrate and stomach cancer (Vincent et al. 1983; Beresford 1985; NRC 1995).

Nitrate contamination of drinking water was found not to be associated with peripheral lymphocyte sister chromatid exchange frequencies in humans (Kleinjans et al. 1991). Nitrate is not mutagenic in bacteria and mammalian cells *in vitro* (Speijers et al. 1989). However, there is some evidence that nitrites or nitro-satiable reaction products are mutagenic in rats, mice and rabbits (Luca et al. 1987).

Furthermore, the results of preliminary studies (Petukhov and Ivanov 1970; Shuval and Gruener 1972; Rotton et al. 1982; Health Canada 1992) indicate that there may be some behavioural effects associated with the ingestion of drinking water with elevated nitrate concentrations; this aspect warrants further study.

5.2. Nitrogen Toxicity in Canadian Environments

Aquatic Ecosystems

Ammonia is not routinely found in Canadian surface waters at high enough concentrations to pose a wide-scale toxic threat to invertebrates or fish. Total ammonia in Canadian surface waters rarely exceeds 0.2 mg NH_3 /L and is usually below 0.1 mg NH_3 /L (Environment Canada 1997b). From 1980 to 1984, total ammonia concentrations in surface waters ranged from 0.002 to 2.00 mg/L in 2 175 samples from across Canada. The highest values were obtained in the Prairies (Environment Canada 1997b). Similarly, 92% of 63 Canadian public water supplies surveyed between 1966 and 1974 contained total ammonia at concentrations of 0.2 mg/L or less (NAQUADAT 1976). Elevated ammonia concentrations are usually associated with mixing zones near municipal and sewage discharges, or with manure or fertilizer runoff from agricultural fields. These high ammonia concentrations are usually dissipated within 10s or 100s of metres from the source because of dilution in the receiving water, conversion of ammonia to nitrate (by nitrification), and volatilization of ammonia to the atmosphere.

Because of the well-documented toxic effects of high ammonia concentrations, discharges of municipal and industrial wastewater in Canada have long been regulated to avoid acute toxicity to aquatic organisms in receiving waters. With respect to agriculture, most Canadian provinces provide recommendations for minimizing nutrient losses to the environment (e.g., Manitoba Agriculture 1994; Brisbin 1995; Brenton and Mellish 1996; NB Agriculture and Rural Development 1997; SAF 1997; AAFRD 1998; MEFQ 1998b; Ontario Farm Environmental Coalition 1998). Nevertheless, there are still incidents of fish kills related to the discharge of N-containing material and, more recently, concern has arisen regarding the role of nitrate in contributing to a decline in amphibian numbers (see Amphibians in Danger text box).

In Canada, 353 fish kills were reported to the National Environmental Emergencies Centre (NEEC) of Environment Canada from 1987 to 1997, of which 22 were caused by discharge of nutrient-containing materials (Table 5.7). It is important to note these incidents are initial pollution reports (unpublished data from NEEC 1998). The incidents are not verified at a later date and only the information actually recorded at the time of the first report is recorded on the pollution report sheets so there may be missing information for some incidents (C. Lau, DOE, personal communication). The nutrient-related fish kills were mainly caused by discharge of N-containing substances; only one incident was related to a phosphorus-containing material and it was likely not the P but another chemical in the product that resulted in toxicity. The majority of the reported fish kills were associated with the agricultural sector, particularly the release of manure. All but one of the fish kills associated with manufacturing and food processing industries were due to the routine discharge of ammonium or ammonia. Fish kills caused by municipal sewage discharge were largely due to equipment failure, discharges from combined sewers, or sewage treatment plant bypasses. A more thorough examination of fish kills caused by agricultural activity in Ontario documented 53 fish kills for a similar time period (1988 to 1998) and showed that most agriculture-related fish kills are caused by spray irrigation of liquid manure from swine operations (see the Fish Kills in Southwestern Ontario case study).

Table 5.7. Fish kills caused by the discharge of nutrient-related material from 1987 to 1997 as reported to the National Environmental Emergencies Centre, Environment Canada, Ottawa. Note that where there has been no finding of fact by a court, the incidents set out in this table are alleged only.

Sector	Material	Number of Incidents	Province
Agriculture	manure	9	ON, MB
	nitrogen fertilizer	1	ON
Chemical Manufacture	anhydrous ammonia/ ammonium chloride	2	ON
Food Processing	ammonia	2	BC, NS
	manure	1	ON
General Manufacturing	concentrated shampoo	1	ON
Municipal Waste	sewage	6	NB, BC, ON

Nitrate is a less serious environmental problem. Although it can be found in relatively high concentrations in surface waters, it is relatively non-toxic to aquatic plants, benthic invertebrates and fish (Russo 1985; Table 5.2). Lethal concentrations of nitrate for a number of amphibian species range from 13 to 40 mg NO_3^--N/L with chronic effects occurring at concentrations as low as 2.5 mg NO_3^--N/L (Baker and Waights 1993; Baker and Waights 1994; Hecnar 1995; Watt and Oldham 1995).

Nitrite is not considered a severe environmental problem because, although extremely toxic to aquatic life, it does not usually occur in natural surface water systems at concentrations considered deleterious to aquatic organisms (Russo 1985; CCREM 1987).

Forest Ecosystems

Concerns about threats to forest health from air pollution led the Canadian Forest Service to establish the Acid Rain National Early Warning System (ARNEWS) in 1984. This national biomonitoring network was designed to detect early signs of the effects of acid rain on forests to enable action to forestall anticipated damage. The term 'acid rain' encompasses all forms of air pollution: wet and dry deposition of sulfates (SOx), nitrogen oxides (NOx), gaseous pollutants (ozone), and airborne particles.

The latest (1994) analysis of data from ARNEWS concluded there was no evidence of a large-scale decline in the health of Canadian forests (NRCan 1999). Tree mortality was mostly in the normal range of 1 to 2% and largely due to natural factors. Nevertheless, certain instances of forest damage were observed where pollutants might be involved. For example, in the Bay of Fundy area of New Brunswick, dieback on birch is coincident with the presence of acid fog and high levels of tropospheric ozone (Cox et al. 1996). Needle flecking has been observed on conifers in Nova Scotia and New Brunswick; its cause has yet to be verified. However, there is no evidence to suggest that elevated concentrations of nitrogen oxides or ammonia or direct deposition of N on vegetation have caused the decline of Canada's forests.

Amphibians in Danger: Are nitrate levels hampering their development?

Recently, it has become evident that amphibian populations around the world have suffered serious decline. In Canada alone, the population numbers for 17 of the country's 24 frog and toad species and 21 salamander species have declined considerably (Jacobs 1999). Because of the worldwide nature of this phenomenon, causes such as acid precipitation, increased ultraviolet radiation, global climate change, or the introduction of exotic species have been put forward as possible explanations (Wyman 1990; Blaustein et al. 1994). Changes in local conditions may, however, offer a better explanation. One possible reason for the declining numbers of amphibians in agricultural landscapes may be the death of their larvae caused by chemical pollutants such as nitrate salts (e.g., Baker and Waights 1993; Hecnar 1995; Marco and Blaustein 1999; Rouse et al. 1999). As well, some of their defining characteristics (permeable skin, dependence on aquatic habitat for reproduction, larval development, forage, or hibernation) contribute to their susceptibility to water pollution (Vitt et al. 1990).

Surface waters naturally contain < 4.4 mg NO_3^-/L; concentrations above 22 mg NO_3^-/L reflect anthropogenic contamination (OME 1981). In North America, nitrate concentrations in agricultural streams typically range between 9 and 178 mg NO_3^-/L (Goolsby et al. 1991; Keeney and Deluca 1993), but levels > 444 mg NO_3^-/L have been found (Rouse et al. 1999). In areas of intensive agriculture in Southern Ontario, nitrate concentrations commonly exceed the Canadian drinking water guideline of 45 mg/L nitrate (Hendry et al. 1983; OME 1983; CCREM 1987; Agriculture Canada 1992). In the laboratory, lethal and sublethal toxic effects are detected for many amphibian species at nitrate concentrations as low as 11-44 mg/L (see Section 5.1). If nitrate in the field acts on amphibians in a similar way to that determined in laboratory studies, then nitrate concentrations in surface waters in North America are very likely adversely affecting amphibian growth and survival (Rouse et al. 1999). Recent studies in the Great Lakes Basin have shown that nitrate concentrations in about 19% of surface water samples were high enough to cause developmental anomalies and 3% were high enough to kill amphibians in laboratory experiments (Rouse et al. 1999).

The primary anthropogenic sources of nitrogenous water contamination in agricultural areas are N-based fertilizers and animal wastes (Goolsby et al. 1991). Nitrogen-based fertilizers have been implicated in the deaths of amphibians in Denmark (Wederkinch 1988), Poland (Berger 1989), Britain (Oldham et al. 1997) and Canada (Hecnar 1995). Amphibian exposure to agriculturally derived nitrate occurs in many ways. In temperate North America, nitrate concentrations in water are usually highest from late fall to spring because of reduced nitrate assimilation by plants (Goolsby et al. 1991; Mason 1991; Rouse et al. 1999). This peak coincides with the period of adult frog hibernation, which leads to prolonged nitrate exposure for adult frogs, especially if they hibernate in water (Hecnar 1995). The type of fertilizer applied is also important. For example, ammonium nitrate fertilizer granules have toxic effects on frogs (*Rana temporaria*) at application rates of 4 to 7 g/m^2 (Oldham et al. 1997), which overlaps with the average application rate (6-9 g/m^2) in Canada (Statistics Canada 1997a; Agriculture and Agri-Food Canada 1998). Under normal application conditions, there is a low probability of an acute impact of ammonium nitrate on adult frogs as granules dissolve rapidly, thereby decreasing the exposure period. Furthermore, the frog's exposure is limited as fertilizer application generally occurs during daylight and most amphibian migration occurs during darkness (Oldham et al. 1997). However, under drier conditions, the toxic effects appear to be longer lasting (Oldham et al. 1997).

Advances in farming methods may also influence exposure to fertilizer. Recently, there has been a shift toward the spring as the peak period for fertilizer application (Chalmers et al. 1990). During this period, chemicals may runoff into ponds and ditches in snowmelt water. In the Prairies, spring application of fertilizer occurs after snowmelt has taken place, so fall applications are more susceptible to enter watercourses via snowmelt (B. Todd, Manitoba Agriculture and Food, personal communication). As frogs commonly migrate over agricultural land to breeding sites during spring, the risk of exposure increases if this movement coincides with a recent fertilizer application (Oldham et al. 1997). This time period also corresponds with an amphibian's vulnerable egg and tadpole stages (Hecnar 1995).

Although nitrate pollution is widespread, the toxicity of this compound is not the only cause of amphibian decline in North America. Other suggested causes include ultraviolet irradiation; genetic mutation; microbes; parasites and other diseases; and other environmental contaminants (Schmidt 1997; Kaiser 1999; Sessions et al. 1999). Nevertheless, water quality data from Canadian agricultural and urban areas suggest that concentrations of nitrate in surface waters exceed toxic levels for amphibians for extended periods of time and during sensitive periods of egg and tadpole development (Rouse et al. 1999).

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Ground Water

Approximately 26% of Canadians rely on ground water for their domestic water supply (Table 5.9). In 1981, this percentage varied from less than 1% in the Northwest Territories to 100% for Prince Edward Island. Ninety percent or more of rural domestic water supplies came from ground water in all provinces except British Columbia and the Yukon and Northwest Territories. Assuming 26% of today's population of more than 30 million people rely on ground water, about 8 million Canadians use it for their drinking water supply. Ground water is also expected to provide a significant percentage of livestock drinking-water supplies in Canada.

Under natural conditions, nitrate-N concentrations in ground water are usually less than 3 mg/L; values greater than this are usually associated with human activity (Henry and Meneley 1993; Carmichael et al. 1995). However, nitrate-N values up to 250 mg/L have been documented in semi-arid regions of Alberta, Saskatchewan, Manitoba and the western United States (Rodvang et al. 1998).

Nitrate occurs in ground water that does not receive a large amount of recharge from precipitation, as indicated by a lack of tritium and high concentrations of dissolved solids. Naturally high nitrate concentrations require the presence of a thick, oxidized, fine-textured deposit where ground water flow has not been sufficient to flush the nitrate (Rodvang et al. 1998). The naturally high concentrations may also be found in areas where native prairie grassland has been tilled. On the Canadian Prairies, the native grassland that formed the basis of the Chernozemic soils was a nitrogen-conserving system and thus, under the natural grass condition, nitrate was seldom present in ground waters of this region. However, when the land was tilled, this nitrate was released from the fibrous root system of the native prairie plants and leached to the ground water (Henry and Meneley 1993). Although nitrate was released as a result of tillage, this nitrate source is still considered to be natural because the nitrate was produced by natural mineralization in the soil.

Nitrate is the most detected ground water pollutant for which a federal drinking water guideline exists (Keeney 1987). Regional surveys of nitrate contamination in ground water wells in Canada indicate that from 1.5 to more than 60% have nitrate-N concentrations greater than the acceptable drinking-water concentration of 10 mg/L nitrate-N (Table 5.10). Aquifers particularly vulnerable to nitrate contamination are shallow (< 30 m below the Earth's surface) and/or situated in very sandy, porous soils (Figure 5.1). Many of these are located in the prime agricultural regions of Canada.

Aquifers in south-coastal British Columbia underlie areas of high rainfall and intensive agriculture and often contain nitrate levels above the safe limit (Table 5.10). Application of manure from intensive livestock operations and extensive fertilization of fruit and vegetable crops are the main contributors of nitrate, although septic systems are also suspected of contributing to the loading in shallow highly permeable aquifers in the region. Sampling of ground water quality for 192 community wells and 75 selected private wells in the Fraser Valley showed that 23 wells (9.6%) had nitrate-N values in excess of 10 mg/L during winter (November 1992 to February 1993) compared to 25 wells (10.1%) during summer1993 (Carmichael et al. 1995). Nearly all of the exceedances occurred in three heavily developed, highly vulnerable, unconfined sand and gravel aquifers: the Abbotsford/Sumas, Hopington and Langley/Brookswood aquifers (see the Abbotsford Aquifer Case Study). Highly vulnerable aquifers are a source of water supply to 50% of the community wells in the study area. During summer 1993, 21 of the 25 wells with nitrate-N over 10 mg/L were private wells.

Agricultural Water Pollution: Fish Kills in Southwestern Ontario

Natural watercourses and drainage ditches are abundant in the fertile ecoregion of the Mixedwood Plains ecozone of southern Ontario. Each year, manure spills occur in this region threatening the health of streams, rivers and lakes by killing fish and other aquatic life, degrading water quality and altering fish habitat. In addition, excessive levels of bacteria have led to concerns about the quality of recreational areas (beach closings) and the use of water by downstream farmers for livestock watering. To assess the impact of agriculture on watercourses in Ontario, data were compiled through the federally-initiated Livestock Manure Pollution Prevention Project on date of manure spill; township and county of spill; fish kill (yes or no); type of operation; route to water; and equipment used.

Data collected between 1988 and 1998 showed that 274 manure spills were reported in Ontario, ranging from 17 to 35 spills per year. Fifty-three of these spills resulted in fish kills, caused either by ammonia toxicity, high inputs of oxygen-demanding material, or both. Of the 274 manure spills, 214 occurred in Southwestern Ontario, particularly in counties with the largest number of intensive livestock operations, and resulted in 42 fish kills (Table 5.8). The majority of spills resulted from spray irrigation of liquid manure from swine operations (41% of all manure spills in Southwestern Ontario), followed by inadequately sized storage facilities (16%) and equipment failure (15%). Most (59%) of the manure spills entered the stream through a tile drain.

Table 5.8. Manure spill data for Southwestern Ontario, 1988 to 1998.

Year	**Number of Spills**	**Fish Kills**	**Route to Stream**		**How Spill Occurred**				
			Field Tile	Overland Runoff	Storage Related	Irrigation Applied	Tanker Applied	Equipment Failure	Transport Related
1988	23	6	21	2	1	21	2	2	2
1989	19	2	8	4	3	6	1	6	1
1990	29	2	22	5	3	15	5	3	2
1991	21	1	12	4	2	10	3	5	2
1992	22	4	11	6	1	11	2	3	2
1993	14	3	9	3	4	4	-	-	-
1994	15	2	9	3	3	2	3	4	2
1995	15	6	5	3	3	2		3	2
1996	20	4	7	2	-	3	2	4	3
1997	23	7	14	12	10	7	4	-	-
1998	13	5	9	4	4	6	2	1	1
Total	**214**	**42**	**127**	**50**	**34**	**87**	**24**	**31**	**17**

Overall the data indicate very few spills were truly accidental. Although causes were related to manure storage, spreading or transportation, the fundamental reason spills occurred was mismanagement. Most of the spills might have been prevented had existing best management practices been implemented, for example:

- where excessive manure was applied, split applications and weather and soil conditions should have been considered;
- if adequate storage capacity had been available in instances where storage facilities overflowed, there may have been enough flexibility to consider weather and soil conditions during application; and
- where equipment failure was a factor, proper maintenance and monitoring of equipment may have prevented spills.

Additional effort is needed to ensure that environmental farm planning, nutrient management planning, manure management planning and contingency planning are in place to minimize manure spills and their environmental impacts.

K. Tuininga, Environment Canada
M. Blackie, Ontario Ministry of Environment

Table 5.9. Population dependent on ground water in Canada. 1981 data from Hess (1986); 1997 data from Federal-Provincial Subcommittee on Drinking Water (M. Giddings, HC, personal communication).

	1981		1997	
	No. of people dependent on ground water	**Percentage of population**	**No. of people dependent on ground water**	**Percentage of population**
Rural population	4 010 000	17	3 million	11
Municipal population	2 210 000	9	3 to 4.8 million	11 to 17.6
Total	6 220 000	26	6 to 7.8 million	22 to 29

The higher number of exceedances for private wells is likely due to the fact that private wells are biased toward high-risk areas and many are located in rural areas where agriculture is more intensive and residents rely on septic systems. Significant nitrate leaching into ground water may also occur in an aquifer near the town of Osoyoos in the lower Okanagan Valley. About 17% of the wells sampled in 1986 and 33% of those sampled in 1987 contained nitrate-N levels higher than 10 mg/L. Fertilizer used in intensive, irrigated tree-fruit production is likely the main source of the nitrate (Reynolds et al. 1995).

The Prairies are considered to have a low risk of contamination due to a dry climate, the generally clayey texture of the soils, and a low cropping intensity. However, there are exceptions, particularly on irrigated lands and those areas near intensive livestock operations that receive large amounts of animal manure. Nitrates due to natural causes have also been reported at concentrations up to several hundred mg/L but these occurrences are highly localized and near the water table (Hendry et al. 1984; Rodvang et al. 1998). In Alberta, about 4 to 6% of domestic wells have nitrate-N > 10 mg/L (Henry and Meneley 1993; Fitzgerald et al. 1997). The low percentage of contaminated Alberta wells may be due to the fact that there are fewer shallow, saturated, surface aquifers in Alberta and that, in many parts of the province, bedrock aquifers are the preferred water source because of their softer water. In Saskatchewan, nitrates exceeded the 10 mg/L nitrate-N guideline in 66 of 184 samples (36%) from private domestic wells in shallow unconfined aquifers (Vogelsang and Kent 1997; Table 5.10). Most of the wells with nitrate exceedances had cattle and/or poultry operations near the well.

Although the results may represent a worst cast scenario for ground water quality in the province, the unconfined aquifers are important because they supply approximately 60% of all farm water supplies in Saskatchewan (Vogelsang and Kent 1997). Data from the Saskatchewan Research Council database and the University of Saskatchewan suggest that from 7 to 17% (Table 5.10) of wells may contain nitrate-N > 10 mg/L (Henry and Meneley 1993). In Manitoba, monitoring conducted by the Department of Natural Resources shows that non-point source contamination of ground water is not significant although a small but measurable increase in nitrate concentration (< 3 mg/L nitrate-N) was detected in portions of the Assiniboine Delta Aquifer where irrigated agriculture is practiced (Henry and Meneley 1993).

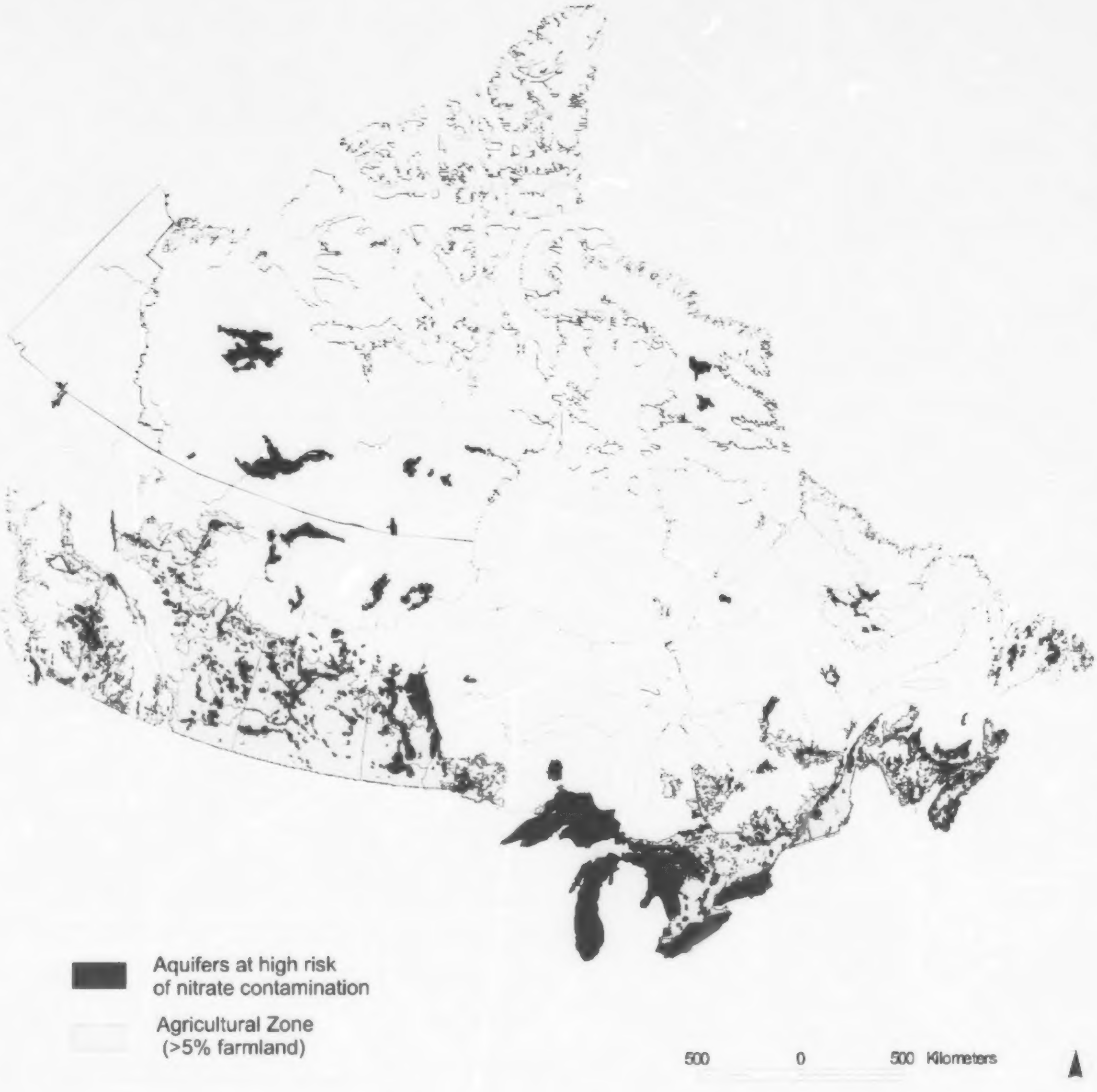
Aquifers at high risk of nitrate contamination
Agricultural Zone (>5% farmland)
500
0
500 Kilometers

Figure 5.1. Canadian aquifers vulnerable to nitrate contamination as identified by Agriculture Canada, 1989 (AC 1989). Vulnerable aquifers are shallow (<30 m below the surface) and/or situated in very sandy, porous soils. Agricultural zone after Smith and McRae (2000).

Table 5.10. Summary of nitrates in rural wells in Canada.

Source of data	No of wells sampled	NO_3-N >10 mg/L (%)	Reference
BRITISH COLUMBIA			
Lower Fraser Valley, Winter 1992-93	240	9.6	Carmichael et al. 1995
Lower Fraser Valley, Summer 1993	238	10.1	Carmichael et al. 1995
ALBERTA			
Alberta Agriculture Database	1425	4.8	Henry and Meneley 1993
Environmental Centre Database	12342	4.3	Henry and Meneley 1993
Alberta Environment Database	1692	3.3	Henry and Meneley 1993
Farmstead Water Quality Survey	813	5.7	Fitzgerald et al. 1997
SASKATCHEWAN			
Saskatchewan Research Council Database	1968	7.2	Henry and Meneley 1993
Soil Salinity Studies	1484	17.0	Henry and Meneley 1993
Shallow Ground Water Quality Survey	184	35.9	Vogelsang and Kent 1997
MANITOBA			
Interlake Carbonate Aquifer	119	1.7	Betcher 1997
Odanah Shale Aquifer	98	19.4	Betcher 1997
Assiniboine Delta Aquifer	29	3.5	Buth et al. 1992
ONTARIO			
Ontario Farm Well Survey, Winter 1991-92	1212	12.8	Goss et al. 1998
Ontario Farm Well Survey, Summer 1992	1212	14.3	Goss et al. 1998
QUÉBEC			
Portneuf	70	41.4	Paradis et al. 1991
Potato Growing Regions	33	63.6	Giroux 1995
Portneuf	26	34.6	Paradis 1997
Montérégie	150	2.0	Gaudreau and Mercier 1997
Orléans Island	87	4.6	Chartrand et al. 1999
NEW BRUNSWICK			
Carleton County	300	11 – 18.2	Ecobichon et al. 1996
Victoria and Madawaska	300	14.5 – 26.1	Ecobichon et al. 1996
NOVA SCOTIA			
Kings County	237	13.0	Briggins and Moerman 1995
PRINCE EDWARD ISLAND			
Water Well Database	2216	1.5	Somers 1998
Dairy Farm Well Survey	146	44.0	Van Leeuwen 1998

In the Odanah Shale Aquifer, however, 19 of 98 samples had nitrate-N above 10 mg/L even though tritium was undetectable in some of the samples indicating naturally-occurring nitrate may be contributing in part to high nitrate concentrations (Betcher 1997).

Nitrate from fertilizer and manure application is a significant contaminant of ground water in Ontario and Québec. A survey of 1 212 domestic farm wells found that 12.8% contained nitrate-N concentrations > 10 mg/L during winter 1991/92 and 14.3% during summer 1992 (Agriculture Canada

1993; Goss et al. 1998; Rudolph et al. 1998) (Table 5.10). Although nitrate contamination in Ontario surveys was not related to any particular land use practice, correlation between contamination of domestic water wells and beneath nearby fields indicates widespread contamination in the aquifer. Application of fertilizer and manure to fields rather than point source contamination from barnyards and septic systems appears to be the source of the nitrate.

Nitrate contamination of ground water in Québec is associated with areas of intensive potato production. Of 70 wells sampled in potato growing areas in Portneuf between 1990 and 1991, average nitrate-N concentrations exceeded the guideline of 10 mg/L for 29 wells (41.4%) (Paradis et al. 1991). A similar study found 21 of 33 domestic wells (63.6%) in potato growing regions of Québec had nitrate-N concentrations > 10 mg/L (Giroux 1995). Investigations in other agricultural and non-agricultural areas found that only three of 150 wells in the Montérégie and four of 87 wells on Ile d'Orléans east of Québec City exceeded the nitrate guideline for drinking water (Gaudreau and Mercier 1997; Chartrand et al. 1999).

In the Atlantic Provinces, ground water contamination occurs mainly in areas where potatoes and corn are produced intensively. Leaching of nitrate can be a serious problem, although levels seem to be steady. Studies of well water quality in agricultural regions of the Saint John River valley, NB (Carleton County, Victoria and Madawaska) in the mid-1980s showed that, depending upon sampling month, up to 22% of wells were contaminated (Table 5.10). There was no correlation of nitrate contamination with distance from fields where fertilizers were used, suggesting that the contamination of ground water was widespread. The presence of nitrate appeared to be closely associated with farming activity as reflected by increasing ground water nitrate concentrations throughout the growing season, particularly following periods of heavy rainfall. Poor well construction also contributed to contamination problems (Ecobichon et al. 1996). In Nova Scotia, nitrate-N concentrations exceeded 10 mg/L for 13% of 237 wells sampled in Kings County, a highly productive agricultural area located at the eastern end of the Annapolis Valley. Point-source contamination from poor well construction and/or maintenance was the primary factor causing contamination (Briggins and Moerman 1995). In Prince Edward Island, 1.5% of 2 216 drinking-water wells had nitrate-N concentrations greater than the 10 mg/L (Somers 1998). A 1997 survey of water samples from 146 dairy farms showed that 44% of the water samples were above 10 mg/L nitrate-N (VanLeeuwen 1998). Monthly monitoring in a suite of 54 wells between 1988 and 1991 indicated that, on average, background concentrations of nitrate from "pristine sites" were three times lower than the average value for the total sample population and four times lower than for samples collected from sites in agricultural areas. Elevated nitrate concentrations were also found in subdivisions with on-site sewage disposal (Somers 1998).

Data on long term changes for nitrate concentrations in ground water in Canada are sparse. A study of ground water age and nitrate distribution within the glacial aquifer used for water supply for the cities of Kitchener-Waterloo showed elevated nitrate concentrations post-1960s, which corresponded to the period when fertilizer application rates increased and corn became a major crop in southern Ontario (Johnston et al. 1998). Of 179 sites with ground water quality monitoring for at least four and up to 25 years in PEI, 33 or 18% of the sites showed increasing nitrate levels, 26.2 or 12%, had decreasing nitrate levels and 124 showed no trend. These results suggest that nitrate levels could reach a steady state for particular types and duration of land use (Somers 1998). Similar experience in Québec indicates that low cost improvements to management practices may lower these levels but levels may still remain above the drinking water guideline for nitrate unless the land use practice is abolished or more costly management practices are adopted.

Abbotsford Aquifer

The Abbotsford-Sumas aquifer is an unconfined, sand and gravel aquifer located southwest of Abbotsford in southwestern British Columbia and northwestern Washington state. It is the largest unconfined aquifer in the Lower Fraser Valley and has an areal extent of approximately 100 km^2 in British Columbia and about 100 km^2 in Washington. The sand and gravel is covered at the surface by a thin soil horizon developed in medium textured eolian deposits.

Elevated nitrate concentrations are observed over a wide area of the aquifer (Liebscher et al. 1992; Carmichael et al. 1995; Zebarth et al. 1998). Using an interpolative technique and ground water nitrate concentrations measured in piezometers and domestic wells in February and August 1993, Wassenaar (1995) estimated that nitrate-N concentrations exceeded 10 mg/L in as much as 80% of the aquifer.

Land use above the aquifer is primarily agricultural with intensive raspberry farming and poultry production accounting for much of the agricultural activity. Urban development is spreading over parts of the aquifer; about 20% of the aquifer surface is covered by urban development (Liebscher et al. 1992). Homes outside municipal limits use septic systems for waste disposal. In many places the septic system tile fields have been installed below the soils in the permeable sand and gravel.

Agricultural production, particularly intensive poultry production, has been suggested as a major contributor to the elevated nitrate concentrations. Poultry manure is frequently applied to raspberry fields as a fertilizer and to enhance the organic content of the soils. Using ^{15}N and ^{18}O isotopes, Wassenaar (1995) concluded that nitrate in the aquifer originated primarily from poultry manure. Fertilizer did not appear to be a significant contributor. Wassenaar (1995) also noted that although nitrate derived from human wastes in domestic septic systems would be indistinguishable isotopically from animal wastes, less than 10% of the nitrate in the aquifer could be derived from septic systems based on the estimated density of septic systems above the aquifer.

Zebarth et al. (1998) used a N balance to estimate N loading from agricultural production to the Abbotsford aquifer. The nutrient balance calculation estimated a surplus of N additions over N removals from the root zone of 245 kg N/ha in 1991. If the same animal and crop management system is used over a long time period and soil N processes reach equilibrium, actual leaching losses should approach the calculated N surplus. Average annual precipitation in the Abbotsford area is about 1500 mm/yr. Approximately 70% of the total precipitation occurs during the October to March period when N removal by crops is low. Because recharge to the aquifer is primarily by infiltration of precipitation, the primary source of elevated nitrate concentrations in the aquifer is likely due to leaching of surplus N from agricultural land over the aquifer.

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Excessive nitrate in ground water was first identified as a health problem when methaemoglobinaemia in an infant was associated with high nitrate well water in Iowa (Comly 1945). Cases from Saskatchewan (Goluboff 1948; Robertson and Riddell 1949), Manitoba and Ontario (Medovy et al. 1947) were subsequently reported in Canada. Worldwide, some 200 cases of infant-methaemoglobinaemia were reported between 1945 and 1970 (Shuval and Gruener 1972). Despite pockets of elevated nitrate concentrations in ground water drinking supplies across Canada, fatalities caused by methaemoglobinaemia are rare (A. Huegenholtz, HC, personal communication). Toxic effects caused by elevated nitrate concentrations are rare because drinking water quality guidelines incorporate a margin of safety, only a small segment of the human population (i.e., infants less than

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Although not common, nitrate poisoning of cattle from well water has been reported in Saskatchewan (Campbell et al. 1954). Sublethal responses resulting in poor animal performance and a general lack of health may not always be recognized, however. The more serious cases of ground water contamination may result in a loss of livestock water supply in rural areas when nitrate-N exceeds the drinking water guideline of 100 mg/L for livestock. The economic costs to replace the contaminated ground water supply may be significant.

5.3. Indirect Toxic Effects

Toxic Algal Blooms

In addition to the direct effects of N on human health, nutrients, especially N, can stimulate growth of toxic species of phytoplankton in both fresh and coastal waters. All phytoplankton species, toxic or not, increase in abundance as a result of nutrient addition however blooms of toxic species are usually most noticed and reported. Consumption of the algal species that produce toxins or the organisms that accumulate algal toxins (e.g., shellfish) can affect the health of terrestrial animals, including humans. Nutrient inputs to inland and coastal waters have increased markedly in recent decades, resulting in changes in phytoplankton growth rates and species dominance (Carmichael et al. 1985; GESAMP 1990; Schramm and Nienhuis 1996; Heinze 1999). Although terrestrial inputs of N (e.g., from runoff and rivers) are the most obvious source of N to oceans, atmospheric deposition is a growing source of N to the North Atlantic Ocean and has been linked to apparent increases in the frequency of harmful algal blooms (Paerl and Whitall 1999).

In marine waters, the most common toxic algae are species of the dinoflagellate family. When certain species of dinoflagellates are present in great abundance a red colouration to the water may be seen, a phenomenon known as “red tide”. Certain dinoflagellate species produce toxins that accumulate in shellfish. Although the shellfish are only marginally affected by the toxins, a single clam can accumulate enough toxin to kill a human (Anderson 1994). The first reported “red tide” causing shellfish toxicity in Canada occurred in April 1933 in Nanaimo, British Columbia (Clemens 1935). However, there was no reported correlation between nutrient inputs and shellfish toxicity in this case. Similarly, Prakash and Taylor (1966) reported a fatal outbreak of shellfish toxicity in the Strait of Georgia in British Columbia in June 1965 from cockles. The shellfish toxins originated from a bloom of *Gonyaulax acatenalla* triggered by high vitamin B_{12} levels in an inlet. Despite the lack of relationship between nutrients and blooms in these early cases, there is now evidence of a link between increased nutrients and the density and scale of a bloom, if not the occurrence of a particular algal species (GESAMP 1990; Schramm and Nienhuis 1996) [see Contaminated Mussel case study].

As a result of concern about shellfish contamination from algae and other sources, the federal government of Canada developed the Canadian Shellfish Sanitation Program and the Canadian Shellfish Water Quality Protection Program. Their main aims are to ensure that growing areas for all bivalve molluscan shellfish (e.g., clams, mussels, oysters, whole and roe-on scallops, and other bivalve molluscs) meet approved federal water quality criteria, that pollution sources to these areas are identified, and that all shellfish sold commercially are harvested, transported and processed in an approved manner. Shellfish are routinely tested for phytoplankton toxins that could be a serious threat to human health (Todd 1990; A. Menon, DOE, personal communication).

In fresh water, the algae causing toxic blooms are usually the blue-green algae. In temperate climates like Canada, such blooms usually occur in late summer and early autumn. Although the causes of bloom formation are not fully understood, conditions that appear to favour blue-green algae blooms include: high concentrations of nitrate, ammonia, and/or phosphate; low irradiance; water temperatures between 15 and 30 °C; and pH between 6 and 9 or higher (Skulberg et al. 1984; Reesom et al. 1994). Bloom development is initiated when warm, stratified surface water low in N undergoes P enrichment (Granéli et al. 1990; Nascimento and Azevedo 1999). Many blue-green algae species are capable of fixing atmospheric nitrogen gas in their cells; hence, in waters with low N concentrations, blue-green algae often out compete other algae and become the dominant taxa.

Not all species of blue-green algae form blooms or produce toxins (Carmichael 1994; Health Canada 1998a). Moreover, blue-green algae that can generate toxins do not always do so. The factors controlling toxin production are poorly understood and many of the factors identified as important in the formation of blue-green algal blooms (e.g., nutrient concentrations, water temperature, light, wind etc.) also appear to be important in toxin production (Carmichael 1986; Wicks and Theil 1990; Kotak et al. 1993a; Chorus and Bartram 1999). Once a bloom occurs, its toxicity can vary over time despite stable environmental conditions. In some cases, blue-green algal scum can remain toxic even after drying out on shorelines.

Blue-green algal toxins are either neurotoxic (i.e., affect the nervous system) or hepatotoxic (i.e., affect the liver) (Kotak et al. 1993b). They are released from the algae when the cells age and lyse (Carmichael 1994). Clinical signs of blue-green algal poisoning include nervous derangement, staggering, tremors and severe abdominal pain (Chorus and Bartram 1999). Ingestion of water containing blue-green algal neurotoxins can lead to death of animals in as little as five minutes. Death results from respiratory arrest due to either neuromuscular blockade (Carmichael et al. 1975) or inhibition of cholinesterase (Cook et al. 1989) (Table 5.11). Two genera of blue-green algae are responsible for the majority of neurotoxin-related poisonings: *Anabaena* and *Aphanizomenon*. In contrast to neurotoxins, blue-green algal hepatotoxins are very stable; death results from destruction of the liver and, ultimately, haemorrhagic shock (Kotak et al. 1993a, b) (Table 5.11). Hepatotoxins result in more cases of animal poisoning, both worldwide and in Canada (Kotak et al. 1995). The most common hepatotoxins are microcystins, produced by many species of blue-green algae including *Microcystis*, the species from which the toxin was first identified.

Blue-green algal toxins can cause illness or death in almost any mammal, bird, or fish which ingests enough of the toxic cells or extracellular toxins (Grant and Hughes 1953; Ingram and Prescott 1954; Gorham 1964; Schwimmer and Schwimmer 1968; Carmichael et al. 1985; Codd et al. 1992; Carmichael 1994). The amount of blue-green algal-tainted water needed to kill an animal depends on such factors as the type and amount of toxin produced by the cells, the concentration of the cells, as well as the species, size, sex, and age of the animal (Carmichael 1994; Chorus and Bartram 1999). Globally, only one confirmed case of human death has been attributed to cyanobacteria (Jochimsen et al. 1998). In 1996, 101 of 124 patients in Brazil who underwent dialysis with water containing blue-green algae toxins (microcystins) had acute liver injury and fifty died. The low incidence of human

Contaminated Mussels- A Prince Edward Island tragedy

In November and December 1987, 105 acute cases of human poisoning and three deaths resulted from the consumption of contaminated blue mussels (*Mytilus edulis* L.) from Prince Edward Island (Bates et al. 1989). Although Canadian authorities had dealt with shellfish poisoning for decades (e.g., Clemens 1935), symptoms were unfamiliar and disturbing. In addition to abdominal cramps, vomiting, diarrhoea and disorientation, some patients exhibited permanent short-term memory loss (Anderson 1994). Fittingly, this new outbreak of shellfish poisoning was termed amnesic shellfish poisoning.

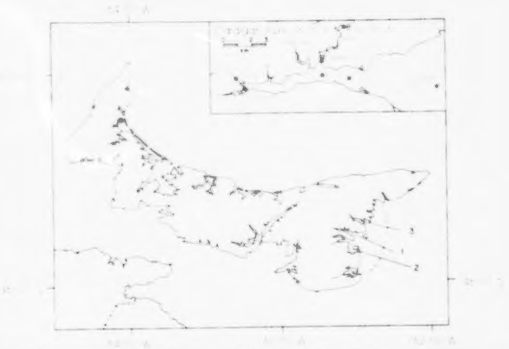

Figure 5.2. Sites where high concentrations of both *Nitszchia pungens* and domoic acid were found in Prince Edward Island in 1987. (1) Cardigan River; (2) St. Marys River; (3) Boughten River. Adapted from Bates et al. (1988).

Commercial mussel cultivation has operated since 1975 on Prince Edward Island in sheltered small river estuaries where salinity is 23-29 parts per thousand and water temperatures vary from -2°C in January to 23°C in late summer (Todd 1990). The origin of the 1987 shellfish poisoning outbreak was traced to the Cardigan River in eastern PEI (Environment Canada and PEI DOE 1990; Figure 5.2). Agriculture forms a major part of PEI's economy and the Cardigan River watershed is no exception. The main farming activities centre on production of livestock (dairy and beef cattle, hogs and foxes); tobacco; grains; potatoes; and food for livestock. As well, an aquaculture site exists on the Seal River, a tributary of the Cardigan River, and there are two land-based finfish aquaculture operations in the watershed (Environment Canada and PEI DOE 1990).

In any outbreak of food poisoning, establishing the poisonous food component and its source is often a difficult task. In mid December 1987, the toxin was identified as domoic acid (Wright et al. 1989). Mussels containing measurable levels of domoic acid were found to be restricted to the Cardigan Bay region (Bates et al. 1991). Domoic acid was not detected in mussels collected in summer and autumn of 1987; however, appreciable amounts were present in mussels collected from the affected area in mid to late October 1987 and reached maximum values in November to December 1987 (Bates et al. 1988; 1991 Figure 5.3). The total amount of domoic acid in all cultivated mussels in the affected area during the 1987 incident was estimated to be 6 kg (Wright et al. 1989). The only obvious potential source of this much domoic acid was the intense bloom of phytoplankton, dominated by the pennate diatom *Nitszchia pungens* (Bates et al. 1988; 1991; Figure 5.3). In mid December 1987, when first sampled, the bloom extended about 10 km seaward from the mouth of the Cardigan River.

The factors influencing the production of domoic acid by *N. pungens* are still not well understood. *N. pungens*, grown in culture, has been shown to produce and release domoic acid with N in the growth medium being essential for domoic acid production (Bates et al. 1991). Subsequently, the abundance of *N. pungens* was correlated with nitrate concentrations in the Cardigan River (Smith et al. 1990). When nitrate was not detected in the water column there was no *N. pungens* bloom. However, when nitrate concentrations in the water column exceeded 1.1 µg/L, a bloom of *N. pungens* developed. The mechanisms producing pulses of nitrate in the Cardigan River estuary were less clear. Rain or wind velocity or both appeared to be involved (Smith et al. 1990), with the most likely nitrate source being freshwater runoff from the agricultural land in the Cardigan

Contaminated Mussels concluded

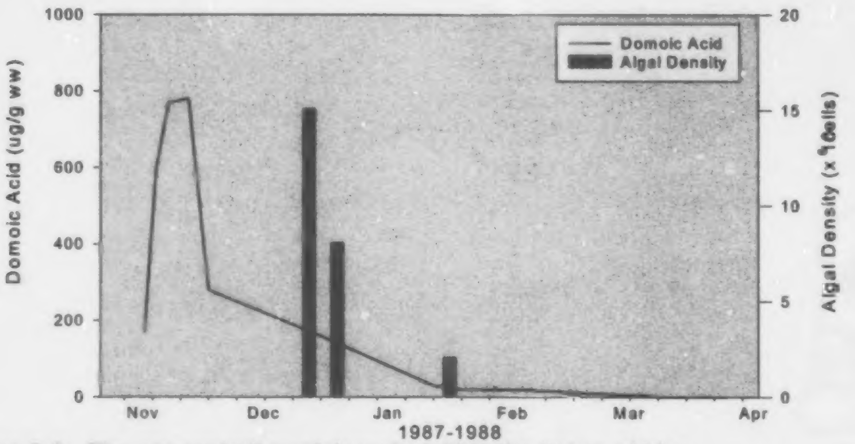

Figure 5.3. The concentration of domoic acid in the tissue of blue mussels (*Mytilus edulis* L.) from Cardigan Bay, PEI, and the density of the pennate diatom *Nitszchia pungens* during the period of mussel toxicity in 1987-1988. Adapted from Bates et al. 1989.

River basin following a rainstorm. Other hypotheses put forward were the suspension of bottom sediments by strong winds and associated release of nitrate, or upwelling of deep nutrient-rich waters. However, bottom sediments are an unlikely source of nitrates and, in the case of upwelling, there was no relationship between the direction of the wind and the timing of the nitrate pulses, nor was there any indication of intrusion of cold deep water. It is therefore thought that the massive bloom of *N. pungens* was the result of a long dry summer followed by nitrate runoff during an intensely wet (but sunny) fall. (J. Wright, DOE, personal communication).

This shellfish incident was unique in Canada in that it was the first known occurrence of human intoxication due to ingestion of domoic acid. As a result shellfish containing more than 20 µg domoic acid/ g shellfish meat is considered unsafe for human consumption (Hallegraeff 1993). The 1987 shellfish incident was a serious threat to the molluscan aquaculture industry in PEI. The establishment of a monitoring program by Department of Fisheries and Oceans and Department of Environment, however, has provided an early warning of toxic events that have protected the industry and consumers of shellfish since 1988.

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Environment Canada

death from cyanobacterial toxins is due not to greater resistance but to an aversion by humans to drink water with algal scum (Health Canada 1998b). Serious human poisonings have, however, occurred after accidental ingestion of toxic cells and water during recreational activities (Kotak et al. 1993b; Lambert et al. 1994; Health Canada 1998b). Blue-green algal toxins can also cause gastroenteritis and contact irritation in users (Billings 1981). Lifetime consumption of water containing low concentrations of blue-green algal toxins may also contribute to the development of cancer (Falconer et al. 1983; Carmichael 1989; Falconer and Buckley 1989; Falconer 1991; Carmichael 1994). Research is currently underway to investigate this link.

Contaminated Mussels concluded

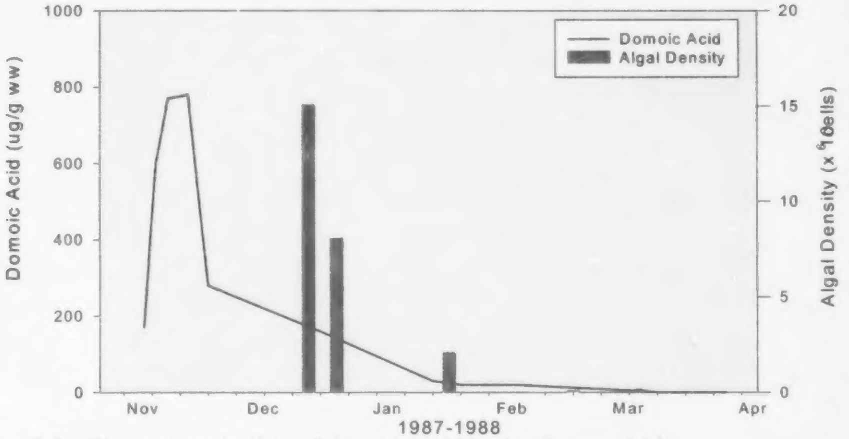

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Table 5.11. Acute and chronic effects of cyanobacterial toxins on terrestrial organisms (IP = administered by intraperitoneal injection; Oral = administered orally; SD = single dose; LD_{50} = the dose of toxins at which 50% of the test organisms died); EC = the effective concentration; NOEC = No observed effect concentration, and MDL_{100} = the minimum lethal dose.

Organism	Endpoint	Conc. (mg/kg)	Exposure period	Comments	Reference
***Microcystis aeruginosa* - toxic isolate-UV-006**					
Mouse	LD_{50}	32570	SD	pH= 6.5; IP	Eloff & Van Der Westhuizen 1981
Mouse	LD_{50}	51470	SD	pH= 8.0; IP	Eloff & Van Der Westhuizen 1981
Mouse	LD_{50}	36200	SD	pH= 10.5; IP	Eloff & Van Der Westhuizen 1981
***Microcystis aeruginosa* - extract of unidentified toxins**					
Mouse	MLD_{100}	40000	60 min.	IP	Østensvik and Skulberg 1981
Mouse	LD_{50}	9100	--	IP	Falconer 1991
Mouse	LD_{50}	28500	24 h	IP	Falconer et al. 1994
Mouse	LD_{50}	1700	--	IP	Falconer et al. 1988
Mouse	LD_{50}	9000	23 day	IP	Falconer et al. 1988
Mouse	LD_{50}	560000	23 day	Oral	Falconer et al. 1988
Mouse	LD_{50}	656000	--	Oral	Falconer 1991
Mouse	LD_{50}	2.6g/kg/d	21 day	Oral	Falconer et al. 1994
Pig	Liver function tests	796.0 & 1312.0 μg toxin/kg/d	71 day	Oral	Falconer et al. 1994
Rat	MLD_{100}	40000.0	3-4 h	IP	Østensvik and Skulberg 1981
***Microcystis aeruginosa* - hepatotoxin- toxic peptide**					
Mouse	LD_{50}	56.0	SD	IP	Runnegar and Falconer 1981
***Microcystis aeruginosa* - microcystin-LR**					
Amphibians (Axolotl; Smooth newt; Marsh Frog)	NOEC (survival); feeding; weight; limb development;	5 and 50 μg/L	SD	Oral to larvae	Oberemm et al. 1999
Fish (Zebra fish; Rainbow trout; Chub)	survival rates; timing of hatching; growth rate; and body length	50 μg/L	SD	Oral to embryos. No acute toxic effects	Oberemm et al. 1999
Mouse	mortality; body weight; alterations in food/water intake	35 μg/mouse	20 μg/L daily for 18 mon.	Oral. No chronic toxicity or accumulation of toxin	Ueno et al. 1999
Mouse	LD_{50}	50.0	SD: 25, 50, or 100 μg/kg toxin	IP	Kotak et al. 1993a
Mouse	LD_{50}	50.0	SD	IP	Carmichael 1988
Mouse	LD_{50}	50.0	SD	IP	Stoner et al. 1989
Mouse	LD_{50}	50.0	SD	IP	Fawell et al. 1994
Mouse	LD_{50}	70.0-120.0	SD	IP	Carmichael 1988
Mouse	LD_{50}	125.0	SD	IP	Brooks and Codd 1987
Mouse	LD_{50}	67.0	SD: 33, 50, 75, or 113 μg/kg toxin	IP	Tsujl et al. 1994
Mouse	LD_{50}	90.0	SD: 40, 60, 90, 135, or 203 μg/kg toxin	IP	Tsujl et al. 1994
Mouse	LD_{50}	5000.0	SD	Oral	Fawell et al. 1994

Table 5.11. Acute and chronic effects of cyanobacterial toxins on terrestrial organisms (concluded).

Organism	Endpoint	Conc. (mg/kg)	Exposure period	Comments	Reference
Rat	body weight; organ weight	50-150 µg/kg/d	28 d	Oral in drinking water	Heinze 1999
***Anabaena flos-aquae* - extract of unidentified toxins**					
Mouse	LD_{50}	26000.0	SD	IP	Falconer 1991
Mouse	LD_{50}	37.5	2 hour	IP	Cook et al. 1989
***Anabaena flos-aquae* - anatoxin-a**					
Mouse	LD_{50}	200	SD	IP	Carmichael 1988
Fish	heart rate	400 µg/L	SD	Oral to embryos. Effects were temporary; no chronic effects	Oberemm et al. 1999
Mouse	LD_{50}	50	SD	IP	Carmichael 1988
***Aphanizomenon flos-aquae* – aphanatoxin**					
mouse	LD_{50}	10	SD	IP	Carmichael 1988

Toxic blue-green algae have been found in lakes in Alberta, Saskatchewan, Manitoba, and Ontario (Gorham 1964; Carmichael et al. 1985; CCREM 1987; Jones et al. 1998). In one Saskatchewan lake, several calves were believed to be poisoned by algae (*Anabaena*) in 1965 and more than twenty dogs between 1960 and 1970 (Carmichael and Gorham 1978; Mahmood and Carmichael 1987). In June 1962, incidents of horse and cow poisonings from toxic algae occurred in Alberta (Gorham 1964). In Manitoba, animal deaths (1 horse; 9 dogs) from algae ingestion have been reported as early as 1951 (McLeod and Bondar 1952). More recent surveys for Microcystin toxins in several provinces showed that during summer 1998, only 28 of 279 raw water samples collected from various surface water sources (lakes, rivers, dugouts, etc.) tested positive (0.3 µg/L to 14 µg/L) using the protein phosphatase method for toxin analysis (Giddings 2000). In 1999, 27 of 92 samples collected tested positive (0.3 µg/L to 9.4 ug/L) for the toxin. The most common algal species identified in these surveys were *Anabaena*, *Aphanizomenon*, *Microcystis* and *Phomidium*.

Toxic algae have also been implicated for direct poisoning of waterfowl populations as well as contributing to outbreaks of avian botulism (Ted Leighton, Canadian Cooperative Wildlife Heath Centre, personal communication). In Saskatchewan, human illness has also been reported from drinking water contaminated with toxic algae; symptoms included stomach cramps, vomiting, diarrhoea, fever, headache, pains in muscles and joints, and weakness (Dillenberg and Dehenl 1960). Algal toxin poisonings of animals in Canadian prairie lakes have generally been associated with the genera *Microcystis*, *Anabaena*, and *Aphanizomenon*. Toxin-producing blue-green algae may also cause economic problems for the aquaculture industry because of fish deaths (Gorham and Carmichael 1980; Skulberg et al. 1984).

At present, the World Health Organization has a provisional guideline of 1 µg/L for microcystin-LR in drinking water (WHO 1997b). Health Canada has proposed a drinking water guideline of 1.5 µg/L for total microcystins based on the toxicity of microcystin-LR (Health Canada 1998a). This proposed guideline is higher than the toxin levels measured for raw water samples (< 0.1- 1.0 µg/L) and treated water samples (<0.1- 0.6 µg/L) in Manitoba in 1995 (Jones et al. 1998) and the rest of Canada (Health Canada 1998a). As well, the Canadian Council of Ministers of the Environment advises against watering livestock from lakes, ponds, or streams that contain heavy growth of cyanobacteria (CCREM 1987).

The best way to avoid the problems associated with blue-green algal blooms is to prevent blooms from forming. Although there is no fail-safe method for doing so, bloom formation can be minimized by controlling nutrient inputs to surface waters and, in the case of reservoirs, ensuring mixing of the water column (Health Canada 1998b). Water supplies contaminated with toxic blue-green algae are often treated with copper-based algaecides (Edmonson 1969; Hodson et al. 1979). However, death of the algae may result in the release of toxins into the surrounding water (Carmichael 1994; Jones and Orr 1994). Human injuries by hepatotoxins have been reported immediately following treatment of drinking water supply with copper sulfate (Bourke et al. 1983).

Trihalomethanes

Chlorination by-products are the chemicals formed when the chlorine used to kill disease-causing micro-organisms reacts with naturally occurring organic matter (e.g., decay products of vegetation) in the water. Nutrient loading can affect the production of chlorination by-products because nutrient addition enhances production of organic matter, such as algae and aquatic weeds, in drinking water supplies. The most common chlorination by-products found in Canadian drinking water supplies are the trihalomethanes (THMs) (Health Canada 1993). The principal THMs are chloroform, bromodichloromethane, chlorodibromomethane and bromoform. Other less common chlorination by-products include the haloacetic acids and haloacetonitriles.

The amount of THMs formed in drinking water can be influenced by a number of factors, including the season and the source of the water. For example, THM concentrations are generally lower in winter than in summer, because concentrations of natural organic matter are lower and less chlorine is required to disinfect at colder temperatures. THM levels are also low when wells or large lakes are used as the drinking water source, because organic matter concentrations are generally low in these sources. The opposite -- high organic matter concentrations and high THM levels -- is true when rivers or other surface waters are used as a source of the drinking water. The preferred approach of controlling THMs is removal of organic matter (the precusors) from the source water before disinfection so that it cannot react with chlorine or other disinfectants to form by-products.

Laboratory animals exposed to very high levels of THMs have shown increased incidences of cancer (see Mills et al. 1998 for a review of health risks). Also, several studies of cancer incidence in human populations have reported associations between long-term exposure to high levels of chlorination by-products and an increased risk of certain types of cancer. For example, a recent study conducted in the Great Lakes basin reported an increased risk of bladder and possibly colon cancer in people who drank chlorinated surface water for 35 years or more. Possible relationships between exposure to high levels of THMs and adverse reproductive effects in humans have also been examined recently. In a California study, pregnant women who consumed large amounts of tap water containing elevated levels of THMs were found to have an increased risk of spontaneous abortion. The available studies on health effects do not provide conclusive proof of a relationship between exposure to THMs and cancer or reproductive effects, but indicate the need for further research to confirm their results and to assess the potential health effects of chlorination by-products other than THMs. Current evidence indicates that the benefits of chlorinating our drinking water -- reduced incidence of water-borne diseases -- are much greater than the risks of health effects from THMs.

5.4. Conclusions

Effects of elevated concentrations of ammonia, nitrate, nitrogen dioxide and nitrite have been studied, with varying degrees of effort, on a diversity of aquatic and terrestrial biota. Ammonia, in the form of un-ionized ammonia, is toxic to aquatic animals (fish and benthic invertebrates) and in most surface waters, total ammonia concentrations greater than about 2 mg/L are toxic. Aquatic plants also show evidence of ammonia toxicity, although they appear more tolerant than aquatic animals. Ammonium (NH_4^+) and un-ionized ammonia (NH_3) in the atmosphere can be toxic to terrestrial plants, whereas inhalation of un-ionized ammonia can be dangerous to humans. In contrast to ammonia, nitrate is essentially non-toxic to aquatic and terrestrial plants. However ingestion of more than 10 mg/L nitrate-N can be lethal for humans; nitrate-N concentrations greater than 1% dry weight in feed can cause death of livestock. Nitrogen oxides in the atmosphere can be toxic to terrestrial vegetation. Comparatively little is known about the toxicity of nitrite because it rapidly oxidizes to nitrate and is rarely present, therefore, in high concentrations naturally. Phosphorus is not directly toxic in its naturally occurring states, even at high concentrations.

Acute lethalities caused by point-source loading of N are uncommon in Canada. This fact is due, in part, to restrictions imposed upon municipal authorities and industries to ensure that wastewater discharges are appropriately treated and diluted to safeguard aquatic ecosystems and the humans that use these waters. Industrial and sewage discharge exceedences are generally unusual events caused by equipment failure or, in the case of sewage discharges, high rainfall or snowmelt that exceeds the capacity of the treatment plant. In contrast to point-source inputs, spills of agricultural N-containing material (particularly manure) are common and often result in fish kills (on average, four reported fish kills per year in Southwestern Ontario, with likely many more unreported kills), suggesting the need for better management of farm nutrients.

In contrast with surface waters where toxic N concentrations are uncommon, concentrations of nitrate in ground water exceed the drinking-water guideline of 10 mg N/L in many parts of Canada. Lethalities are avoided as a result of public education programs advising parents who use ground water as their drinking-water source to make up formulas for infants with bottled water. Livestock lethalities resulting from consumption of nitrate-enriched plants are also avoided by education of livestock producers.

The chronic effects of elevated concentrations of ammonia, nitrate, nitrogen dioxide and nitrite are less well understood. Because the symptoms of chronic nitrogen toxicity may mimic or be masked by other environmental changes, the extent of chronic nitrogen toxicity in the environment is generally not known. However, recent evidence suggests that nitrate concentrations in runoff and surface waters in southern Ontario are sufficiently high to cause chronic (as well as acute) nitrate toxicity in amphibians. Evidence is accumulating that suggests the decline of amphibians, such as in southern Ontario, may be due, in part, to toxic concentrations of nitrate.

In addition to the direct toxic effects of certain N compounds, nutrient addition to fresh and coastal waters enhances the growth of algae, including species of algae that produce toxins. Consumption of algal species that produce toxins (in the case of fresh water) or the organisms that accumulate algal toxins (in the case of coastal waters) can affect the health of terrestrial animals, including humans. In Canada, toxic algal blooms occur most frequently in lakes and potholes in the Prairies. Here, deaths of livestock and pets are reported each year. Toxic algal blooms have occurred on both the Pacific and

Atlantic coasts of Canada; however, their frequency is sporadic in coastal waters. In response, in part, to the 105 cases of poisoning and three deaths caused in 1987 by consumption of mussels containing an algal toxin, the Government of Canada developed the Canadian Shellfish Sanitation Program and the Canadian Shellfish Water Quality Protection Program to ensure that growing areas for all bivalve molluscan shellfish meet approved water quality criteria.

Nutrient loading can also indirectly affect the quality production of treated drinking water supplies. Chlorine addition to disinfect water can lead to the formation of a number of chlorination by-products if the water supply contains organic matter such as algae or aquatic weeds. Among the many disinfection by-products, trihalomethanes (THMs) are most often present. The preferred approach of controlling THMs is by removing organic matter (the precusors) from the source water before disinfection.

F3

6.0 Scientific Rationale for the Management of Anthropogenic Nutrient Sources

Highlights

- In Canada, excessive nutrient loading was first recognized as a problem in the Great Lakes in the late 1960s.
- Federally, the *Canadian Environmental Protection Act* is the only regulation that addresses phosphorus and then only in laundry detergents. It prohibits the manufacture, use, and sale of laundry detergents that contain P in concentrations exceeding 2.2 % by weight.
- Water Quality Guidelines in Canada are developed to provide basic scientific information about water quality parameters and to protect Canadian species and water uses. Most provinces have developed their own water quality guidelines for various nutrients (i.e., nitrate, ammonia, phosphorus), but in the absence of provincial guidelines, Canadian Water Quality Guidelines are usually applied.
- Varying degrees of nutrient legislation, regulations, objectives, guidelines, best management practices (BMPs) and other measures exist in all provinces and territories.
- Many of the approaches used to minimize nutrient loading and associated environmental impacts in Canada are also employed in other jurisdictions. However, compared to Canada, countries with a longer history of settlement and agriculture have tended to adopt tighter restrictions to minimize nutrient loading to the environment.
- New technologies for reducing nutrient impacts to the environment continue to be developed. Environmental policy should continue to emphasize integrating the best and most advanced science into practical solutions.

Nutrients are managed or are of concern in many countries around the world. As long ago as the 16th century, controls were implemented in parts of Europe to restrict the disposal of household waste, largely in an attempt to limit plagues and other health concerns caused by bacteria or the pests attracted to refuse. By the 1800s, disease outbreaks caused by bacterial contamination of water used for drinking or in food preparation were prompting engineering solutions, such as wastewater diversion, to solve water pollution problems. At present, access to clean water in relatively abundant quantities is taken for granted by most people in developed countries. However, many people do not realize that their actions put a burden on water quantity and quality – from water used in the production of produce, meat and dairy products for human consumption, to water used in industrial processes for cooling and cleaning, to tap water used for cooking, cleaning and personal hygiene. To protect the natural environment, to safeguard human health and to preserve water quantity and quality for beneficial uses such as drinking, industrial water supply, and recreational pursuits, nutrient management measures have been adopted in Canada and elsewhere.

This chapter identifies the various nutrient management programs operating in Canada, at both the federal and the provincial/territorial levels, and discusses the rationale behind these programs. The extent to which scientific concerns and issues have played a role in developing nutrient control measures in Canada is also examined. Nutrient control measures for other countries are also discussed, with a particular emphasis on measures not routinely employed in Canada. Finally, new or emerging technologies are described that may aid in reducing nutrient loading to the environment or minimizing impacts.

6.1. Current Regulations, Guidelines, and Best Management Practices in Canada

In Canada, the first problem of accelerated eutrophication due to excessive P inputs was identified in the late 1960s for the Great Lakes. Although P and other nutrients occur naturally in the Great Lakes, excessive amounts of anthropogenically derived P led to severe disruptions in the ecosystem (see Lake Erie text box, Chapter 4.2). Concern by the governments of both Canada and the USA that the huge Great Lakes system could be so affected by anthropogenic nutrient loads led to the signing of the Canada-U.S. Great Lakes Water Quality Agreement (GLWQA), an agreement aimed at reducing P loading to the Great Lakes.

Since that time, other environmental problems caused or associated with anthropogenic nutrient additions have been identified (Table 6.1). Governments (federal and provincial/territorial) as well as citizens' groups, industrial associations, agricultural producers and other concerned parties have responded with programs ranging from community-based volunteer programs to legally enforceable legislation (Tables 6.1 and 6.2).

This section focuses on initiatives undertaken in Canada in response to environmental concerns caused by nutrient addition from human activity. Existing control measures, including legislation (Acts and Regulations), guidelines, objectives, and voluntary initiatives that govern nutrients, fertilizers, and eutrophication are reviewed. The rationale behind these controls is identified, including an overview of any water or land use impairment, impact or perceived impact resulting from nutrients and fertilizers. Finally, additional programs aimed at assessment and remediation of the impact of nutrients and fertilizers are evaluated. The information contained in this section is based largely on interviews with individuals working in a variety of environmental and agricultural ministries across Canada. The anecdotal comments supplied by the interviewees provide an historical context and general overview of nutrient issues in various Canadian jurisdictions.

Federal Initiatives

Canada-United States Great Lakes Water Quality Agreement

Public concern over excess algae and plant growth in the lower Great Lakes, particularly Lake Erie, provoked investigation into probable causes. Scientific studies concluded that an excess of P from detergents and other cleaning products, municipal and industrial effluents, and agricultural runoff resulted in increased plant life which, in turn, led to increased organic material for decomposition and lower oxygen concentrations.

Various proactive measures, beginning with the 1972 signing of the *Canada-U.S. Great Lakes Water Quality Agreement*, have attempted to work towards a solution to the P problem. The Agreement includes municipal and industrial commitments to remove P from effluents and to eliminate P from detergents (Table 6.1). Canada's commitment to survey and monitor the Great Lakes under the *Canada-U.S. Great Lakes Water Quality Agreement* involves analyzing water samples for nutrients, organic contaminants, and physical parameters. These samples are collected as part of the Great Lakes Surveillance Program. The Surveillance Program, begun in 1967, focuses on one lake each

Table 6.1. Identified concerns and issues regarding nutrient control measures at the federal and provincial/ territorial level and the subsequent responses of each jurisdiction (Information up to June 2000.).

Jurisdiction	Identified Concern/ Issue	Response
Federal	• Eutrophication of the Great Lakes	• *Canada Water Act* (1973) - help reduce P loading into Canadian lakes • *Phosphorus Concentration Control Regulations* (1989) - limits P in cleaning products
	• Canada & U.S. concern about P in effluent and eutrophication of Great Lakes	• *Canada-U.S. Great Lakes Water Quality Agreement* (1972) - commitments to remove P from detergents and effluents • Surveillance and monitoring of the Great Lakes to evaluate trends and identify emerging issues
	• Environmental problems caused by toxic substances, hazardous wastes, and pollution of air and water	• *CEPA* (1999) - focus on assessment and regulation of toxic substances and hazardous waste emphasizing pollution prevention, ecosystem approach, biodiversity, the precautionary principle and user-producer responsibility
	• Requests for regulation and control of agricultural fertilizers	• *Fertilizers Act* (1885) - regulates the safety, efficacy and labelling of fertilizers imported into or sold in Canada
National	• Inconsistent and incomplete water quality standards	• *Canadian Water Quality Guidelines* (1999) and *Drinking Water Guidelines* – development of general, consistent, and scientifically sound management tools to ensure societal stresses do not lead to the degradation of Canadian waters (including drinking water supply; freshwater and marine life; agricultural uses; recreation and aesthetics; and industrial uses)
Alberta	• Complaints from residents regarding excessive algae, degraded water quality, and impaired water use	• Studies on the Bow River, in combination with other studies examining P and N controls, led the City of Calgary to voluntarily implement P controls at Calgary's WWTPs in 1982-83 • Developed partnerships with interested parties/ residents and stakeholders (e.g., Pine Lake Restoration Program)
	• Disposal of sewage sludge	• Guidelines for the Application of Municipal Wastewater Sludges to Agricultural Lands (1997) - developed to allow monitoring and controlled application to agricultural lands
	• Intensive livestock operations and their impact on the environment	• Codes of Practice for the Economic and Safe Handling of Animal Manures are voluntary guidelines for producers to manage agricultural nutrients in manures
	• Recognition that agriculture can have a deleterious effect on aquatic systems	• Industry-led Beneficial Management Practices manual development
	• P enrichment of surface waters in Alberta	• Development of P soil limits for application of manure on agricultural lands
British Columbia	• Eutrophication in various water bodies in the province (i.e., Okanagan Basin)	• *Pollution Control Objectives for Municipal Type Waste Discharges in British Columbia (1975)* - permissible limits for discharged nutrients • *Waste Management Act* (1996) - nutrient discharge limits based on site-specific studies
	• Environmental impacts of agricultural wastes	• Agricultural Waste Control Regulation (1992) and Code of Agricultural Practice for Waste Management - practices for using, storing, and managing agricultural wastes • Management of Livestock and Poultry Manures in the Lower Fraser Valley (1994-1997) - joint provincial/ federal study • Tackling Non-Point Source Water Pollution - document compiled by MELP examining the association between agriculture and other activities and non-point source water pollution and providing best management practices.

Table 6.1. Identified concerns and issues regarding nutrient control measures (continued).

Jurisdiction	Identified Concern/ Issue	Response
Manitoba	• Prevention of unacceptable water quality impairment • Excessive growth of algae and macrophytes and overall eutrophication of surface waters	• *Manitoba Surface Water Quality Objectives* (1988) - conducted site-specific studies and developed water objectives • *Manitoba Clean Water Guide* (1997) - public education initiative and material • *Livestock Manure and Mortalities Management Regulation* (1998) - developed to control nutrient contamination • Much of the Department of Environment's resources are directed at dealing with this issue on a continuous basis
New Brunswick	• Water resource impacts from land use activities	• Promotion of Best Management Practices (BMPs) and development of recommended agricultural practices by the Department of Environment • Watershed groups have initiated a number of activities and projects to increase awareness about nutrient issues
Newfoundland	• Eutrophication of water resources related to agricultural seepage and surface runoff	• Promotion of good agricultural practices through provision of educational materials (technical information on nutrients and fertilizers) • *Farm Practice Guidelines for Livestock & Poultry* - currently under development
Northwest Territories	• Preserve the state of the North's waters and avert the problems experienced in southern Canada	• *Guidelines for the Discharge of Treated Municipal Wastewater in the Northwest Territories* (1992) - limits for P • *Receiving Water Quality Objectives* - recommend nutrient levels to avoid nuisance growth
Nova Scotia	• Nutrient related impacts on a site-specific, local scale • Environmental impacts related to agricultural practices	• NSDOE supports local programs and community-based partnership initiatives at specific sites in the province • Development of guidelines for land-use activities (having non-point sources of nutrients) such as forestry and agricultural operations • Development of educational materials related to residential development • Municipal wastewater approvals for larger operations require receiving water studies which can include TP modelling and associated limits • Introduced a Green Farm Initiative, involving BMPs to remediate and prevent environmental impacts to water resources
Ontario	• Maintain a certain level of quality in Ontario's surface waters • Undesirable changes in the aquatic system from excessive algal growth (partly due to recreational development of lakes) and watersheds of high P concentrations due to agricultural inputs and low flow • Eutrophication of the Great Lakes in the 1960s and obligations under the Canada-U.S. *Great Lakes Water Quality Agreement* (through the Canada Ontario Agreement of 1971)	• *Water Management Policies, Guidelines, Provincial Water Quality Objectives(PWQOs) of the Ministry of Environment and Energy* (1994) - direct management of waters • Interim PWQOs for P - to be used as a guideline (in conjunction with other site-specific programs) to protect against aesthetic deterioration; eliminate excessive plant growth in rivers and streams; avoid nuisance concentrations of algae; and protect the diversity of trophic status • Led to Certificates of Approval to limit loadings of P from individual dischargers into receiving waters
Prince Edward Island	• Eutrophication and excessive plant production	• Roundtable on Resources & Sustainable Land Use (1997) - reported on the impact of land uses on water resources • Developing legislation on water buffer zones • Voluntary programs initiated to clean up water quality problems

Table 6.1. Identified concerns and issues regarding nutrient control measures (concluded).

Jurisdiction	Identified Concern/ Issue	Response
Québec	Environmental problems caused by toxic substances released in water	Programme d'assainissement des eaux du Québec (1970s) - water clean up program for urban, industrial, and agricultural sectors
	Maintain a certain level of quality in Québec's surface waters	*Québec Surface Water Quality Guidelines / Critères de qualité de l'eau de surface au Québec* (1998) - direct management of waters
	Eutrophication of water resources related to intense agricultural activities (i.e., P and seasonally NH_3).	Regulations to reduce pollution from agricultural activities / *Règlementations sur la réduction de la pollution d'origine agricole* (1997) - information provided on nutrients and fertilizers to promote agricultural practices • Developing legislation on water buffer zones. • Directives for manure storage
Saskatchewan	Excessive aquatic weed and algae growth in the Qu'Appelle River System	Prairie Provinces Water Board Objectives (1969) - implemented to address nutrients, fertilizers, and eutrophication Review of nutrient loadings to the Qu'Appelle River and Fishing Lakes in the mid 1970's recommended further nutrient control measures
	Impact of nutrient releases on Saskatchewan watersheds	Development of ecological monitoring parameters by the Ecosystem Management Special Projects Unit
Yukon	Nutrients are generally not an issue	Domestic sewage from most communities is treated, retained in lagoons, and then discharged to lands or wetlands

year (except Lake Michigan, as it lies entirely within the United States), with Upper and Lower Lake surveillance cruises alternated on a biannual basis. The Program's objectives are to ensure compliance with water quality objectives, evaluate trends and identify emerging issues.

Data from the Ontario Water Resources Commission Sampling Program have also been useful in measuring the response of the nearshore Great Lakes to the International Phosphorus Control Program (Nicholls et al. 1980) and are essential to fulfilment of terms outlined by the *Canada-U.S. Great Lakes Water Quality Agreement*. This ongoing program, now managed by the Ontario Ministry of Environment, is part of an integrated plan involving several collaborators with the ultimate objective of generating data to assess the dynamics of the Great Lakes and interrelations among nutrients, organic contaminants, and biota.

Fertilizers Act

Improper application of fertilizers can result in nutrients leaching away from crop roots and into surface and ground waters. The *Fertilizers Act* is product-based legislation regulating fertilizers and supplements imported into or sold in Canada (Table 6.1). This legislation deals only with the actual product and its associated proprieties (composition, including formulants, contaminants, and sources of active ingredients; efficacy; safety in use; and representation in the marketplace, specifically through labelling).

The potential for fertilizers to cause eutrophication is addressed, in a general way, by the *Fertilizers Act* through labelling requirements. Recommended rates of application are developed at the provincial level by agricultural ministries, and use of a product is not recommended if eutrophication could result. Misuse of fertilizer products, resulting in damage to the environment, is also typically the responsibility of provincial ministries of the environment.

Fisheries Act

The 1985 *Fisheries Act* provides broad information for the protection of all fish habitat (Table 6.1). When the *Fisheries Act* was introduced, it was designed to address the basic concerns for coal ash and ballast (oil); there is nothing specific in it with respect to nutrients. However, Section 36 of the *Fisheries Act* prohibits the release of any substance that could have a deleterious impact on fish and their habitat and this allows for the inclusion of nutrients as deleterious substances. Regulations of the *Fisheries Act* generally regulate point sources, and are thus unlikely to address non-point nutrient sources such as agricultural runoff.

As Fisheries and Oceans Canada promotes aquaculture, they may be faced with concerns such as eutrophication and the impact of biocides and toxicological products designed to prevent disease. Although there is a general sense that water quality is a provincial concern, there is an understanding that in the future, there may be a need for some regulation of aquaculture activities under the *Fisheries Act*. Addressing the impact of nutrients would be an important aspect for consideration in future aquaculture regulations.

Canadian Environmental Protection Act

The 1988 *Canadian Environmental Protection Act* (CEPA) was created with the intention of responding to environmental pollution caused by toxic substances and hazardous wastes (Table 6.1). It incorporated provisions in the *Canada Water Act* authorizing regulation of P content in detergents and water conditioners. The renewed *CEPA* passed in 1999 emphasizes pollution prevention and the consideration of a holistic ecosystem approach when addressing environmental contamination. Under *CEPA*, the Governor in Council may, on the recommendation of the Minister, make regulations "for the purpose of preventing or reducing the growth of aquatic vegetation that is caused by the release of nutrients in waters and that can interfere with the functioning of an ecosystem or degrade or alter, or form part of a process of degrading or altering, an ecosystem to an extent that is detrimental to its use by humans, animals or plants" (*CEPA* 1999 Part 7, Division 1, 118(1)).

To date, federal nutrient regulations made under *CEPA* have focused solely on controlling P in laundry detergents. The 1989 *Phosphorus Concentration Regulations* prohibit the manufacture, use, and sale of laundry detergents containing P in concentrations exceeding the maximum permissible limit of 2.2 % by weight.

National Initiatives

Guidelines for Canadian Drinking Water Quality

The *Guidelines for Canadian Drinking Water Quality* identify substances found in drinking water and known, or suspected, to be harmful. For each substance, the Guidelines establish the maximum acceptable concentration (MAC) that can be permitted in water used for drinking. MAC values are typically 10 to 5 000 times lower than those levels at which any adverse effects on health have been observed during prolonged and repeated testing. The Guidelines also define a number of desirable aesthetic attributes of drinking water -- qualities that give it a pleasing appearance, taste and odour. Canadian drinking water guidelines have been developed for a variety of microbiological, chemical, physical and radiological parameters. These guidelines, which apply to drinking water from all private and municipal water sources, are developed by the Federal-Provincial Subcommittee on Drinking Water and are published by Health Canada. The Subcommittee is composed of members from all provincial and territorial governments, as well as a member from Health Canada and Environment Canada. Health Canada acts as scientific advisor to this committee.

The Guidelines are recognized throughout Canada as the standard of drinking water quality. They provide a convenient, reliable, yardstick against which drinking water quality can be measured. Provision of drinking water is, however, a provincial responsibility; the guidelines are not applied federally except in those areas that fall under federal jurisdiction. Virtually all provincial and territorial governments have established their own measures of drinking water quality based on the Guidelines.

Guidelines for Canadian Recreational Water Quality

The *Guidelines for Canadian Recreational Water Quality* deal with health hazards associated with recreational water use, as well as aesthetic and nuisance conditions. Recreational waters refer to those natural waters used not only for primary contact activities such as swimming, windsurfing, and water-skiing, but also for secondary contact activities such as boating and fishing. Recreational use is defined as any activity involving intentional immersion (e.g., swimming) or incidental immersion (e.g., water-skiing) of the body, including the head in natural water. Natural water is in turn defined as any marine, estuarine or fresh body of water, as well as any artificially constructed flow-through impoundment using untreated natural water. The health hazards associated with direct contact with water include infections transmitted by pathogenic organisms as well as injuries and illness due to physical and chemical properties of the water.

The Guidelines are developed by an *ad hoc* Working Group formed at the request of the Federal-Provincial Committee on Environmental and Occupational Health (CEOH). The guidelines are periodically revised or adjusted as new or more significant data become available and are not to be regarded as legally enforceable standards except where promulgated by the appropriate provincial or federal agency.

Guidelines for Canadian Water Quality

The Canadian Council of Ministers of the Environment (CCME) was requested to develop nationally applicable guidelines by various organizations and jurisdictions in Canada. The original approach in the development of national guidelines was to attempt to adopt and modify existing guidelines from other sources. However, in many cases, existing guidelines proved to be inappropriate or unavailable. From this original attempt at guideline development, the need for nationally consistent, scientifically defensible standards for freshwater and marine life, and human drinking water was identified. As a result of the CCME's initiatives, the *Canadian Water Quality Guidelines* (CWQGs) were developed (CCME 1999; Table 6.1). References to the *Guidelines for Canadian Drinking Water Quality* and the *Guidelines for Recreational Water Quality* are reprinted in CCME's CWQGs with permission from Health Canada as both of these documents are published under the authority of the Minister of Health.

The rationale for these Guidelines was to provide basic scientific information on the effects of water quality variables on Canadian waters. The guidelines were developed to ensure that anthropogenic stressors, particularly toxic chemicals, do not lead to degradation of raw water for drinking water or industrial supply; water inhabited by freshwater or marine life; water used for agricultural purposes; or water used for recreation and appreciated for its aesthetics. The Guidelines may be either a numeric value or a narrative statement. When several water uses are to be protected at a certain location, the Water Quality Guideline to be applied for a specific substance is that which provides protection for all users (i.e., highest and best use). This requirement means that the most restrictive guideline value is selected, and by default, it protects all users. For the majority of substances, aquatic life guideline values are usually the most restrictive (CCME 1999).

The CWQGs address concerns regarding increased plant growth caused by excessive nutrients. Eutrophication is addressed specifically in the Recreational Water Quality (Health Canada) and Aesthetics Section, where guidelines for avoidance of nuisance aquatic weeds growth are contained. Many of the provinces/territories have adopted the nationally derived CWQGs. However, some jurisdictions have developed their own N and/or P guidelines.

In 1999, the CWQGs were updated based on new scientific information. The new *Environmental Quality Guidelines* (EQGs) were expanded to cover, in addition to water quality, guidelines for air, soil, sediment quality, and tissue residues (CCME 1999). The EQGs include over 550 numerical or narrative guidelines for over 220 substances of concern. In addition, these guidelines offer the most current science on the fate and effects of these substances in the Canadian environment.

Provincial and Territorial Jurisdictions

Alberta

In Alberta, most rivers arise in the mountains and are inherently oligotrophic. However, the quality of these rivers may be impaired as a result of municipal or industrial discharges or agricultural runoff. Alberta's lakes, on the other hand, often have high background nutrient levels and although nuisance algal blooms may be in part a natural phenomenon, development activity has increased nutrient concentrations in some lakes with associated deleterious effects. Concerns about water quality led to the development of provincial water quality guidelines, the most recent version being *Surface Water Quality Guidelines for Use in Alberta* (1999) (Table 6.3).

To address residents' complaints about water quality and impaired water use, the government is building partnerships with interested parties and stakeholders (Table 6.1). For example, residents, farmers, and business people of Pine Lake, AB raised funds to begin clean up to protect the lake. In partnership with Alberta Environment, they commissioned monitoring studies, implemented a device to pump nutrient-laden water from the bottom of the lake, and initiated several beneficial management practices (BMPs) on farming operations.

Nutrient impacts on surface waters have also led to the adoption of nutrient controls for MWTPs in large cities (Table 6.1). An example, the Oldman Water Quality Initiative, has led to improved water quality in the Oldman River basin following the 1999 implementation of tertiary treatment with P removal at the City of Lethbridge municipal wastewater treatment plant. Improved sewage treatment results in great production of sewage sludge. To ensure appropriate use of sewage sludge as a soil amendment, *Guidelines for the Application of Municipal Wastewater Sludges to Agricultural Lands* (2000) were developed. Some smaller communities discharge wastewater to lagoons for long term storage that can subsequently be used for irrigation.

The 1993 *Environmental Protection and Enhancement Act* (*EPEA*) is a broad piece of legislation that regulates substances released into waters in the province of Alberta (Table 6.2). Using regulations, facility design, and water quality stipulations, the *EPEA* has the authority to regulate municipalities for nutrients (municipal and industrial N and P compounds) discharged into the environment. Although livestock operations and their associated nutrients have seldom been regulated under *EPEA*, charges can be laid under various clauses. For example, the clauses dealing with substance release and deposition of manure on ice can be used. The *Code of Practice for the Safe and Economic Handling of Animal Manures* (AAFRD 2000) gives direction for establishing and operating livestock facilities, including recommendations on the application of manure to farmland. Alberta is also considering guidelines for management of intensive livestock operations.

British Columbia

In the 1970s, in response to signs of eutrophication in numerous water bodies, British Columbia developed nutrient objectives (Table 6.1). The objectives, outlined in the *Pollution Control Objectives for Municipal Type Waste Discharges in British Columbia* (1975), are used to set permanent limits for discharged nutrients. Separate *Pollution Control Objectives in British Columbia* were also prepared for the *Chemical and Petroleum Industries* (1974); the *Food-Processing, Agriculturally Orientated and other Miscellaneous Industries* (1975); the *Forest Products Industry* (1977); and the *Mining, Smelting, and Related Industries* (1979). To provide further protection, federal and provincial grants were provided in the 1980s to upgrade many sewage treatment facilities to tertiary treatment processes with biological nutrient removal.

British Columbia is currently in the process of developing municipal sewage regulations to update and replace the existing 1975 *Pollution Control Objectives for Municipal Type Waste Discharges in British Columbia*. Within these proposed regulations, the discharger will examine the nutrient levels in the receiving environment and propose a level of treatment based on this assessment. There are exceptions for freshwater systems of special concern, such as the Okanagan Basin. The best example of a site-specific case study in British Columbia was a major study carried out jointly by provincial and federal governments in the Okanagan Basin. This study found P loading to the basin to be the cause of eutrophication problems. Nutrient discharge limits to these waters have since been set for discharge permits issued under the *Waste Management Act* (1996).

The British Columbia Ministry of Agriculture and Food produces *Environmental Guidelines for Producers* that provide farmers with technical information on nutrients and fertilizer application to promote good practices (BCAF 1999). Also in 1992, the *Agricultural Waste Control Regulation* with its associated *Code of Agricultural Practice for Waste Management* was issued. The purpose of the *Code* was to describe practices for using, storing, and managing agricultural wastes in an environmentally sound manner. Since then, compliance has been attempted by a two-pronged process involving: (1) agricultural peer groups who handle low risk complaints; and (2) Ministry of Environment, Lands, and Parks (MELP) staff who deal with high risk problems. Industry and government have developed other strategies and guidelines to ensure compliance, including commodity producers.

Table 6.2. Legislation, regulations, objectives, guidelines, and associated control measures relevant to nitrogen (N), phosphorus (P) and other nutrient-related issues, as of June 2000. The table also provides details on the history and rationale surrounding these control measures.

Jurisdiction	**Relevant Legislation, Regulations, Objectives, Guidelines, and other Control Measures**
Federal	• *Fertilizers Act* (1885) • *Canada-United States Great Lakes Water Quality Agreement* (1972) • *Canada Water Act* (1973) - developed to help reduce P loads into Canadian lakes; now part of the *Canadian Environmental Protection Act* • *Fisheries Act* (1985) • *Phosphorus Concentration Control Regulations* (1989) - limit for P in cleaning products • *Yukon Waters Act* (1993) - addresses contaminants (i.e., waste discharges) and their effects when added to waters (habitat alteration, adverse effects to humans, fauna and flora) • *Canadian Environmental Protection Act* (1999; originally 1988) – nutrients addressed directly in the regulation for P in cleaning products and indirectly in the *Ambient Air Quality Objectives* (1989) that provide three ranges of ambient air quality objectives for NO_2
National	• *Canadian Water Quality Guidelines* (1987) - nationally applicable, scientifically based management tools • *Canadian Drinking Water Quality Guidelines* (1996) - guidelines for N compounds for the safety of consumers • *Canadian Environmental Quality Guidelines (1999)* – updated *Canadian Water Quality Guidelines* (e.g., nitrate, nitrate, ammonia)
Alberta	• *Guidelines for Limiting Contaminant Emission to the Atmosphere from Fertilizer Plants and Related Industries* (1976) - regulation under the *Clean Air Act* (1975) that suggests guidelines for ammonia and ammonium phosphate emitted into the air from fertilizer plants • Wastewater Effluent Guidelines for Alberta Petroleum Refineries (1985) - wastewaters are monitored for maximum deposits of substances (i.e., chemical oxygen demand and ammonia) to a water course from refineries • *Environmental Protection and Enhancement Act* (1993) - broad legislation that regulates substances released into water • *Alberta Ambient Air Quality Guidelines (1993)* – regulation under the *Environmental Protection and Enhancement Act* to assess air quality and to assist in setting source emission standards in approvals (e.g., NO_2) • *Standards & Guidelines for Municipal Waterworks, Wastewater & Storm Drainage Systems* (1997) - under Best Practicable Technology Standards there is standard for total P for municipalities of a certain size • Surface Water Quality Guidelines for Use in Alberta (1999) - updated Alberta Surface Water Quality Objectives (1977) and Alberta Ambient Surface Water Quality Interim Guidelines (1994) • Wastewater Management Review for Fertilizer Manufacturing Sector (1999) – update for Fertilizer Plant Wastewater Effluent Guidelines (1976). Compares Alberta wastewater management practices with management practices in other jurisdictions to determine technology based wastewater discharge limits (e.g., ammonia) • Water Act (1999) – provides legislative authority for implementing new polices accepted by government to guide water management • Framework for Water Management Planning (2000 draft) – requires the Government to establish a strategy for protecting the aquatic environment • Guidelines for the Application of Municipal Wastewater Sludges to Agricultural Lands (2000) - regulate the application of sludge to monitored lands • Guideline for Municipal Wastewater Irrigation (2000) - ensure that municipal wastewater is used for irrigation only when environmentally acceptable and agriculturally beneficial (e.g., nutrient, salt, sodium contents are examined)

Table 6.2. Legislation, regulations, objectives, guidelines, and associated control measures (continued).

Jurisdiction	**Relevant Legislation, Regulations, Objectives, Guidelines, and other Control Measures**
British Columbia	• *Pollution Control Act* (1967) - based on the combined results of a public inquiry and ecological, health, technological, and economic considerations; it aims to preserve land, water, and air environments at the highest possible levels • *Pollution Control Objectives for Municipal Type Waste Discharges in British Columbia* (1975) - N in effluent may be limited if it controls eutrophication or is abnormally high (based on site-specific studies) • *British Columbia's Water Quality Criteria for Nutrients & Algae* (1985) - protect water resources from degradation and use impairment by associating algae in areas of eutrophication with concentrations of P in order to set limits • *British Columbia's Water Quality Criteria for N (Nitrate, Nitrite, and Ammonia)* (1986) - criteria to control N compounds to protect specific water uses • *Agricultural Waste Control Regulation and Code of Agricultural Practice for Waste Management* (1992) - describes practices for good agricultural practices (i.e., use, storage, and handling of manure) • *Sanitary Regulation* (1992) – prohibits activities which make water unfit for human or domestic purposes • *Production & Use of Compost Regulations* (1993) - Regulation under the *Waste Management Act* that prescribes specific limits for N and P in composting waste • *Waste Management Act* (1996) - discharge limits for nutrients • *Water Act* (1996) • *Fish Protection Act* (1997) - protect and enhance fish habitat, acknowledges nutrient issues, but has no specific control measures for nutrients • Sewage Disposal Regulation (1997) – Regulation designed to ensure consideration of assimilative capacity of the receiving environment through prior treatment of waste • *Tackling Non-Point Source Water Pollution* (1997) - document by MELP examining the association between agriculture and other activities and non-point source water pollution • *Forest Practices Code of British Columbia* (1998) – prescribes land use practices on Crown Forest and Range land for the protection of water quality in community watersheds • *Health Act* (1998) – Allows for the termination and removal of health hazards or potential health hazards
Manitoba	• *Environment Act* (1987-1988) • *Pesticides and Fertilizers Control Act* (1987) - regulates the sale and use of fertilizers in the province • *Guidelines for Various Air Pollutants* (1988) – regulation under the *Environment Act* that provides two levels of ambient air quality and objectives for various air pollutants (e.g., NO_2) • Private Sewage Discharge Systems and Privies Regulation (1988) - no person shall discharge sewage or sewage effluent into or onto the surface or the ground except in compliance with this Regulation • *Anhydrous Ammonia Handling & Transportation Regulation* (1989) • *Manitoba Clean Water Guide* (1997) - educational guide created by Manitoba Environment for use by cottage associations, conservation districts, educational institutions, etc., to increase public awareness and participation regarding water quality issues • *The Livestock Manure and Mortalities Management Regulation* (1989) - one intention of this Regulation is to minimize the potential for excessive nutrient contamination of surface waters • *Manitoba Water Quality Standards, Objectives, and Guidelines* (2000) - limits for P to minimize algae and macrophyte growth and for several N compounds to ensure passage, maintenance, and propagation of fish species; protection of livestock and poultry; and safety of waters for swimming, boating, and aesthetic purposes • *Development of a Nutrient Management Strategy for Surface Waters in Southern Manitoba* (2000 draft) – outlines the main issues that need to be considered and addressed in the development and implementation of a nutrient management strategy
New Brunswick	• *Clean Environment Act* (1973) and *Clean Water Act* (1989) - both serve to control releases into water and to deal with sites where a release has occurred; and to protect qualities of water, animal/ plant life, and enjoyment/ use • Water Quality Regulation (1982) - contains numerical values for numerous parameters that should be met in the province's waters • *Watercourse Alteration Regulation* (1990) - indirectly addresses nutrients and their impacts when introduced into aquatic systems

Table 6.2. Legislation, regulations, objectives, guidelines, and associated control measures (continued).

Jurisdiction	**Relevant Legislation, Regulations, Objectives, Guidelines, and other Control Measures**
New Brunswick	• *Guidelines for Livestock Manure and Waste Management in New Brunswick* (1996) - assess the adequacy of design and operation of all livestock and poultry operations with respect to good agriculture practice and pollution control • *Air Quality Regulation* (1997)– regulation under the Clean Air Act (1997) that limits the release of some air contaminants (e.g., NO_2) so that the ground level concentrations do not exceed the maximum permissible ground level concentrations
Newfoundland	• *Waste Material Disposal Act* (1990) • *Environment Act* (1995) • *Environmental Control Water & Sewage Regulations* (1996) - regulations under the *Environment Act* that give specific values of N and P to control environmental releases • *Air Pollution Control Regulations* (1996) – regulations under the Environment Act that provide air quality standards and emission standards for certain nutrient-related pollutants (e.g., NH_3, NO_2, N_2O) • *Environmental Guidelines for Agricultural Development* (of certain sized livestock & poultry operations); *Guidelines & Conditions for the Approval of Swine, Poultry & Other Livestock Enterprise*; *Information Guide for Livestock Facilities*; *Manure & Waste Management in Newfoundland*; *Urban & Rural Planning Act*; and the *Municipalities Act* all broadly address nutrients • *Farm Practices Guidelines for Livestock and Poultry* - currently under development to deal specifically with nutrients and eutrophication • Environmental Farm Plan Initiative and Safety Net Programs assist livestock and poultry producers in developing manure storage and handling practices
Northwest Territories	• *Environmental Protection Act (1988)* • *Guidelines for the Discharge of Treated Municipal Wastewater in the Northwest Territories* (1992) - qualitative objectives for nutrients (i.e., avoid nuisance conditions) in order to meet the objectives of the NWT Water Board (conservation, development, and utilization of water resources) and to protect human health • *Guidelines for Industrial Waste Discharge into the NWT* (1998) - objectives for BOD, P, and ammonia • *Guideline for Agricultural Waste Management* (1999) – purpose is to establish clear and consistent waste management standards for the intensive livestock and agricultural industry in the NWT
Nova Scotia	• Water Act (1989) • *Environment Act* (1995) • *Air Quality Regulations (1995)* – regulations under the Environment Act providing criteria for ambient air quality for some pollutants (e.g., NO_2), expressed as maximum permissible ground level concentrations • *Water and Wastewater Facility Regulations* (1995) • *Activities Designation Regulations* (1995) • *On-site Sewage Disposal Systems Regulation* (1997) • *Guidelines for Forestry Practices* - on crown lands • *Guidelines for Manure Management* - recommendations regarding manure storage and spreading • Watershed management plans are currently being developed for waters designated under the Canadian Heritage River Program (two rivers in NS) and those in municipal water supply areas. The plans address nutrients and eutrophication • NSDOE issues approvals for sewage treatment plants by setting biological oxygen demand (BOD), total suspended solids (TSS) levels and occasionally ammonia levels, based on assimilative capacity of the receiving environment to receive nutrients (*e.g.*, total P) • Provincial approval for land based aquaculture operations in freshwater environments have set controls, based on the expected efficiency of solids management system, for TSS and total P loading NS is investigating options to better address aquacultural effluents and better protect receiving environments (New Brunswick approach is being investigated) and NS has adopted policy which prohibits commercial finfish aquaculture in lakes. These controls are partly a response to concerns regarding impacts to fish habitats aesthetics and recreational and drinking water uses

Table 6.2. Legislation, regulations, objectives, guidelines, and associated control measures (continued).

Jurisdiction	Relevant Legislation, Regulations, Objectives, Guidelines, and other Control Measures
Nova Scotia	• Metro Areas Lakes Program - community-based program which monitors nutrients and chlorophyll *a* in urban areas • Volunteer Lake Monitoring Program - helps to protect lakes against the impacts of residential development • Green Farm Initiatives - uses BMPs to remediate and prevent environmental impacts to waters

Ontario	• *Environmental Protection Act* (1990) - identifies nutrient issues in the following legislation:
	• *Agricultural Code of Practice for Ontario* (1973) - addresses issues such as areas for manure utilization, handling, and spreading in winter and spring;
	• *Guidelines for the Control of industrial Phosphorus Discharge in Liquid Effluents* (1976);*Discharge of Sewage from Pleasure Boats Regulation* (1990) - regulates the discharge of organic and inorganic wastes;
	• *Guidelines for Sewage Sludge Utilization on Agricultural Lands* (1977 – revised 1986)
	• *Guidelines for the Treatment and Disposal of Liquid Industrial Wastes in Ontario* (1978)
	• *General- Air Pollution Regulation* (1990 – revised 1998) – controls air contaminants (NH_3, NO_2)
	• *Ambient Air Quality Regulation (1990* – revised 1994) – identifies a desirable ambient air quality criteria for some contaminants (e.g., NO_2); and
	• *Guidelines for the Protection and Management of Aquatic Sediment Quality in Ontario* (1992) - protects against harmful levels of contaminants including nutrients (total organic carbon, total Kjeldahl nitrogen, total phosphorus) in sediments
	• *Ontario Water Resources Act* (1990) - broadly addresses nutrients in the following documents:
	  • *Water Management Policies, Guidelines, Provincial Water Quality Objectives* (1994) - manages surface and ground water quality in the province through numerical Provincial Water Quality Objectives (PWQOs). Currently, there is a proposed Interim PWQO for P
	  • Waterworks and Sewage Works Regulations (1993)
	• Municipal Water and Sewage Transfer Act (1997)
	• Nutrient Management Plans - BMPs deal with the over application of nutrients and contain P index for Ontario for both soil and open surface water runoff
	• Provincial Water Quality Monitoring Network collects water quality data, including P measurements, and based on the properties and quality of the receiving waters. Certificates of Approval are issued to individual dischargers to limit P loadings

Prince Edward Island	• *Environmental Protection Act* (1988) - contains information regarding the adverse effects of nutrients • *Air Quality Regulations* (1992) – regulations under the *Environmental Protection Act* that sets out maximum ambient air contaminant levels (e.g., NO_2) based on ground level concentrations standards • Water Quality Certificate Regulation (1995) - issued when water parameters do not exceed the maximum acceptable concentrations or aesthetic objectives, as specified in the current *Guidelines for Canadian Drinking Water Quality* • 1997 Roundtable on Resources and Sustainable Land Use has led to the current development of legislation on buffer zones that will address nutrients and eutrophication indirectly
Québec	• *Programme d'assainissement des eaux du Québec (1970s)* - water clean up program for municipal, industrial, and agricultural wastes • Environment Quality Act (1977) - addresses contaminants (nutrients indirectly) • Bottled Water Regulations (1981) - limit the concentrations of nitrate + nitrites • Drinking Water Regulations (1984) - limit the concentrations of nitrate + nitrite • *Québec Surface Water Quality Guidelines / Critères de qualité de l'eau de surface au Québec* (1998) - values for maximum concentrations of total P and several N compounds • *Reduction of Pollution from Agricultural Sources Regulation / Règlementation sur la réduction de la pollution d'origine agricole* (1997) – agricultural management regulation under the *Environmental Quality Act* addressing the over application of nutrients, livestock waste storage, and spreading of fertilizing substances • Directives for manure management - for the protection of ground water and streams • Legislation on buffer zones - indirectly protects surface waters against direct nutrient releases

Table 6.2. Legislation, regulations, objectives, guidelines, and associated control measures (concluded).

Jurisdiction	Relevant Legislation, Regulations, Objectives, Guidelines, and other Control Measures
Saskatchewan	• Prairie Provinces Water Board Water Quality Objectives (1969) • *Environmental Management and Protection Act* (1983) • *Guidelines for the Use of Sewage Sludge on Agricultural Lands* (1987) - maximum values for both total N and P in sludge to ensure sewage sludge is applied to agricultural lands in a beneficial and environmentally acceptable manner • *Water Pollution Control and Waterworks Regulations* (1989) • *Clean Air Regulations* (1990) – regulations under the *Clean Air Act (1986)* setting ambient air quality standards for certain pollutants (e.g., NO_2)
Yukon Territory	• Environment Act (1991) - nutrients as contaminants are addressed • Contaminated Sites Regulation (1996) - aquatic regulations under the *Environment Act* for aquatic life, livestock watering, drinking water for nitrate and nitrate + nitrite • Sewage discharged to land or wetlands has undergone secondary treatment or better (with the exception of two communities)

One example of action taken on the environmental effects of agricultural waste management was the joint provincial/ federal study, *Management of Livestock and Poultry Manures in the Lower Fraser Valley* (Brisbin 1995). The association between agriculture and other activities and water pollution was also examined by the MELP with findings in the document *Tackling Non-Point Source Water Pollution* (Brisbin 1995). Future actions to mitigate the problem of over-production of manure in the Lower Fraser Valley are being developed through consultations with stakeholders and include the following

- MELP, the Ministry of Agriculture and Food, and three agricultural organizations have developed a Memorandum of Understanding to formalize peer advisory services to assist with resolving environmental issues and to promote sustainable agriculture. A mediation process to resolve operational issues of the services was recently initiated.
- Producer Conservation Organizations were established with assistance from federal Green Plan funding. These developed on-farm stewardship programs.
- The Sustainable Poultry Farmers Group initiated a manure transportation program to assist in achieving compliance with the *Code of Agricultural Practice for Waste Management*. This approach has resulted in approximately 40% of the excess manure in the South Matsqui area being relocated to other areas.
- Government and the hog industry are developing a transition plan for hog operations to achieve compliance. Industry in conjunction with government is examining the feasibility of treatment facilities and expanded storage facilities. There will be some funding available for storage facilities through Investment Agriculture.
- A recently formed Ten-Point Action Plan on Agriculture and the Environment was agreed to by the agriculture industry and the provincial government. A Working Committee is currently reviewing all agricultural-environmental issues requiring consultation and action.

The new 1997 provincial *Fish Protection Act* (Section 12) requires municipal bylaws to be implemented, where directed, regarding the protection and enhancement of riparian areas that may be subject to residential, commercial, or industrial development. Agriculture is exempted from this Section of the *Act* on the condition that other policy tools be developed to protect and enhance riparian areas within agricultural use areas.

Manitoba

To prevent unacceptable levels of water quality impairment, the Manitoba government developed the *Manitoba Water Quality Standards, Objectives, and Guidelines* (2000), which contain specific

Table 6.3. Water quality guidelines in Canada for nutrients (N and P) and aquatic plant biomass. AQL= Aquatic Life; ALL= All Water Uses; REC = Recreation and Aesthetics; LW= Livestock Watering

VARIABLES	VALUE µg/L	USE	RATIONALE	JURISDICTION	REFERENCE
Total P	5 - 15	AQL	Lakes only, with salmonids as predominant fish species [1].	BC	Pommen 1989
	10 (maximum)	REC	Lakes only [1].	BC	Pommen 1989
	10	ALL	For a high level of protection against aesthetic deterioration in lakes naturally below this value; ice-free period.	ON	OME 1984; OMEE 1994
	20 (average)	ALL	To avoid nuisance concentrations of algae in lakes; ice-free period.	ON	OME 1984; OMEE 1994
	< 30	ALL	To eliminate excessive plant growth in rivers and streams.	ON	OME 1984; OMEE 1994
	50% of "predevelopment" value (interim)	ALL	To provide guidance for protecting water quality and clarity.	ON	OME 1998
	20 - 30	ALL	Based on Ontario values (OMOE 1984) to eliminate the nuisance of algae and aquatic plants	QC	MEFQ 1990
	50% of natural ("background") level	ALL	Lakes with P levels between 10 and 20 µg/L; ice-free period. For sensitive habitats (e.g., lakes with trout), criteria will be validated with models predicting hypolimnetic oxygen.	QC	MEFQ 1998a
	50	ALL	Water quality suitable for most uses.	AB	Alberta Environment 1999
Elemental P	0.1	AQL	Based on U.S. values (USEPA 1976); for salted waters.	QC	MEFQ 1990
Total N	10 000	REC	For the prevention of nuisance growths of aquatic plants.	BC	Pommen 1989
	1 000	ALL	For total inorganic and organic N, based on the toxicity of ammonia and nitrate and on the role of N as a nutrient.	AB	Alberta Environment 1999
	1 000	ALL	Multipurpose water quality objective. For total organic and inorganic N based on toxicity tests.	SK	SERM 1983
	Narrative	ALL	For total organic and inorganic N to prevent nuisance growth of aquatic rooted, attached and floating plants, fungi, etc. which would render the water unsuitable for beneficial uses.	MB	Williamson 1983
Nitrate + nitrite (as N)	100 000	LW	To protect livestock when both nitrate and nitrite are present.	BC	Nordin and Pommen 1986
	100 000	LW	To afford protection to most livestock species as well as to the consumers or products derived from these livestock.	SK	SERM 1995
	100 000	LW	To protect livestock when both nitrate and nitrite are present.	MB	Williamson 2000
	100 000	LW	To protect livestock when both nitrate and nitrite are present.	ON	OME 1984
	100 000	LW	Based on USA (NAS/NAE 1973); Ontario (OME 1984) and Manitoba (Williamson 1983).	Canada	CCREM 1987

VARIABLES	VALUE µg/L	USE	RATIONALE	JURISDICTION	REFERENCE
Nitrate (as N)	200 000 (maximum) 40 000 (average)[2]	AQL	To protect freshwater aquatic life, based on literature reports.	BC	Nordin and Pommen 1986
	200 000	AQL	To protect freshwater aquatic life, based on British Columbia values.	QC	MEFQ 1990
	100 000	REC	For waters for which high concentrations of nitrate are not likely to cause any direct problems in terms of body contact or visual deterioration. To protect recreational users who may ingest water, it is recommended that the drinking water criteria be applied.	BC	Nordin and Pommen 1986
	100 000	REC	Based on British Columbia values. To protect users from the risk of ingesting the water, it is recommended to apply the criteria of drinking water to all water that is used for recreation.	QC	MEFQ 1990
	Narrative	ALL	A numerical guideline is not given for nitrate, but concentrations that stimulate prolific weed growth should be avoided	Canada	CCREM 1987
Nitrite (as N)	60 (maximum) 20 (average)	AQL	For the protection of aquatic life, on the basis of literature reports.	BC	Nordin and Pommen 1986
	60	AQL	Based on British Columbia values. A value of < 20 µg/L is reported as a chronic toxicity value.	QC	MEFQ 1990
	60	AQL	Based on toxicity test with fishes; will protect most species.	Canada	CCREM 1987
	10 000	LW	To protect livestock from toxic concentrations of nitrite alone.	MB	Williamson 2000
	10 000	LW	Based on livestock toxicity and Manitoba criteria.	Canada	CCREM 1987
	1 000 (maximum)	REC	To protect all water users against possible ingestion.	BC	Nordin and Pommen 1986
	1 000	REC	Based on British Columbia criteria	QC	MEFQ 1990
Total ammonia (NH_3 + NH_4^+ as N)	680 – 2 770	AQL	To protect freshwater aquatic life; varies with pH (6.5 – 9.0) and temperature (0 – 20°C).	BC	Nordin and Pommen 1986
	600 – 2 060	AQL	To protect freshwater aquatic life; varies with pH and temperature.	SK	SERM 1995
	21 – 231 000	AQL	To protect freshwater aquatic life; varies with pH (6.0 - 9.5) and temperature (0 - 30°C).	Canada	CCME 2000
	2 500 (maximum) 1 000 (30 day mean)	AQL	To protect marine aquatic life	BC	Nordin and Pommen 1986
	102 – 2 770	AQL	To protect marine aquatic life; varies with pH (6.5 - 9.0) and temperature (0 - 20°C); based on British Columbia criteria.	QC	MEFQ 1990
Ammonia (un-ionized – NH_3)	18 – 50	AQL	Varies with pH (6.5 - 9.0) and temperature (0 - 30°C).	MB	Williamson 2000
	19	AQL	Freshwater aquatic life. Based on least observed effects concentration (pathological lesion and tissue degradation in the kidney) in rainbow trout.	Canada	CCME 2000

VARIABLES	VALUE µg/L	USE	RATIONALE	JURISDICTION	REFERENCE
	20	AQL	Based on the lowest concentrations reported for aquatic toxicity, bioaccumulation and mutagenicity.	ON	OMEE 1994
Nutrients	narrative	ALL	N and other nutrient concentrations should not be altered from natural levels by discharges of effluents such that nuisance growths of algae or aquatic weeds result; to provide a minimum degree of protection of all beneficial uses [3].	SK	SERM 1983; SERM 1995
Aquatic plants	narrative	REC	Biota, which could be a nuisance to bathers if present in large numbers, should be absent from areas intended for development of bathing beaches. Biota includes floating or rooted aquatic plants, phytoplankton and periphyton.	Canada	Health and Welfare Canada 1983; CCREM 1987
Attached benthic algae	30 to 50 mg/m^2 chlorophyll *a* (interim)	AQL	The identification of objectionable biomass levels for aesthetic concerns is clearly a subjective matter. However, benthic algae in this range seems to be the "break point: between what is acceptable versus objectionable in salmonid streams.	Canada	Environment Canada 1995
	50 mg/m^2		To protect uses related to recreation and aesthetic in streams	BC	Nordin 1985
	100 mg/m^2		To protect against undesirable changes in aquatic life in streams	BC	Nordin 1985

[1] Refers to the spring overturn concentration (lake epilimnetic residence time greater than 6 months), or the mean epilimnetic growing season concentration (epilimnetic residence time less than 6 months)

[2] The average value is calculated from at least 5 weekly samples taken in a period of 30 days.

[3] These general objectives are applicable to all portions of a receiving waterbody except that portion designated by the Department as an effluent mixing zone. They are not applicable to intermittent streams except during flow periods such as spring runoff. These objectives should be achieved however, in portions of intermittent streams where ponded stream water is utilized for beneficial purposes.

Other Jurisdictions:
PEI: use CCME guidelines.
NF: use CCME guidelines.
NWT: use CCME guidelines.
NB: use CCME guidelines for the most part. Exceptions are the water quality objectives developed by Canada--US--state--provincial committees for the international portions of the Saint John River and the St. Croix River. Because CCME has not yet developed a guideline, NB uses the Ontario guideline of 30 µg/L total P for flowing waters. For lakes, NB usually refers to the P values which Dillon and Rigler (1975) used to define four trophic levels and to predict lake development capacity based on their trophic status.

requirements for nutrients (Table 6.3). These objectives are analogous to the *Canadian Water Quality Guidelines*, but have been adapted for site-specific use. The *Manitoba Water Quality Standards, Objectives, and Guidelines* assist in developing industrial and municipal waste discharge restrictions, assessing surface water quality, and determining the suitability of waters for certain uses. The information provided by the *Manitoba Water Quality Standards, Objectives, and Guidelines* also plays a role in predicting treatment costs at specific locations and in development of strategies to control land use practices.

The province has produced a number of educational materials about water quality issues aimed at increasing public awareness and participation. The 1997 *Manitoba Clean Water Guide*, issued by Manitoba Environment, and several pamphlets on the water quality of specific water bodies are examples of such materials used by cottage associations, conservation districts, and educational institutions.

Studies on specific parameters (i.e., P study of Clearwater Lake) as well as overall water quality assessments (i.e., water quality assessment of Whitemud River Watershed) have been completed by Manitoba Environment. Data from government studies are used in conjunction with information from such volunteer programs as the Manitoba Environment Volunteer Water Sampling Program to guide future management decisions in the province.

Manitoba provides information on practices and systems related to the storage, handling and disposal of livestock wastes (including manure) through its *Farm Practices Guidelines for Dairy (or Poultry, Beef or Hog) Producers in Manitoba* (Manitoba Agriculture 1994). The Manitoba Ministry of Agriculture produces the *Soil Fertility Guide* providing farmers with technical information on nutrients and fertilizer application in order to promote good practices (Manitoba Agriculture 1999).

Manitoba also has guidelines for managing application of sludge or biosolids to land in Manitoba. The concern over nutrients (organic nitrogen, ammonia, nitrate) is addressed in these general guidelines that vary from one location to another (e.g., soil type, sensitivity of groundwater, etc.), and are reflected in legally-binding licenses issued under the *Manitoba Environment Act*. Treatment of sludge prior to application would normally include anaerobic digestion or equivalent treatments. The level of treatment, including anaerobic digestion, will be specified in each license.

A nutrient management strategy is currently being developed for Manitoba, which will include the development of nutrient objectives for rivers and streams on a combination of regional-scale characteristics and considerations and the nutrient carrying capacity of Lake Winnipeg.

New Brunswick

New Brunswick relies, for the most part, on guidelines in the CCME *Canadian Water Quality Guidelines* (1999). However, the Province's 1989 *Clean Water Act* and the 1990 *Watercourse Alteration Regulation* both have authority to address indirectly nutrients introduced to aquatic systems and their subsequent impacts. The Province is currently developing and implementing a *Water Classification Regulation* that will include water quality standards for dissolved oxygen, bacteria, and benthic community structure for flowing waters. During the initial development of the *Clean Water Act*, this action had been considered as a method to address water resources on a watershed basis and to provide a role for the public in setting water quality goals.

For stream protection, New Brunswick currently uses Ontario's guidelines to set limits for nutrients in streams receiving effluent. In addition, New Brunswick promotes best management practices (BMPs) for land use activities. Site-specific BMPs have been developed such as the use of buffer strips to reduce nutrient loading to streams. For example, while the Department of Environment has the authority to specify which practices are permitted within a measured zone around drinking water sources, outside of this zone, they encourage agricultural practices that will maximize protection of watersheds. Watershed groups have also initiated a number of activities and projects in an attempt to increase awareness about nutrient issues. A riparian zone restoration campaign by the Trout Creek Model Watershed Group and septic system upgrades in the Buctouche River Sustainable Development Project are two examples.

New Brunswick also publishes *Manure Management Guidelines for New Brunswick* (NBARD 1997), which give direction for establishing and operating livestock facilities, including the application of manure to farmland.

Newfoundland

In Newfoundland, the protection of water quality, habitat, and the environment from nutrients and other agents are addressed in the 1990 *Waste Material Disposals Act*; the 1995 *Environment Act*; and the 1996 *Environmental Control Water and Sewage Regulation*. In addition, the province is developing *Farm Practice Guidelines for Livestock and Poultry*. These will deal specifically with nutrients, fertilizers, and eutrophication. Also, the Agricultural Extension Services, Specialists on Crops, Soil & Agricultural Engineering Services provides farmers with technical information on nutrients and fertilizer application in order to promote good practices. This initiative aims to prevent eutrophication of water resources associated with agricultural seepage and surface runoff.

Northwest Territories

Although much of the North is not subjected to anthropogenic nutrient loading, municipal wastewater discharges associated with a growing and more urban population have the potential to affect human health and environmental quality. The policy of the Northwest Territories Water Board is to apply best practical means in treating municipal waste before it is discharged into waters. The intent of this policy is to preserve the state of the North's waters and to "avert the problems allowed to build up in southern Canada where often strict controls were not imposed in the early days of municipal growth" (Water Board of NWT 1992).

In terms of nutrients, the *1992 Guidelines for the Discharge of Treated Municipal Wastewater in the Northwest Territories* and the 1998 *Guidelines for Industrial Waste Discharge in the Northwest Territories* recommend limits for P, though they apply only under certain conditions and are rarely a licensing condition. The type of sewage treatment facilities most practical in northern communities was an important consideration in developing these guidelines. *Receiving Water Quality Objectives* address nutrient concerns and recommend that levels outside the initial mixing zone be maintained in such a way as to avoid nuisance conditions.

Nova Scotia

In Nova Scotia the management and protection of water resources are addressed through legislation such as the 1995 *Environment Act*, the 1995 *Activities Designation Regulations*, the 1995 *Water and Wastewater Facility Regulations*, and the 1997 *On-Site Sewage Disposal Regulations*. These do not address nutrients directly but provide the authority to develop site-specific requirements. Site-specific development controls are established to reduce water quality impacts from various activities that can include nutrients sources. Predictive modelling and assessments of assimilative capacity have been used in establishing limits.

Operational guidelines have been developed for various land-use activities (e.g., forestry and agriculture) which promote the use of BMPS and reduction of impacts from non-point sources. Sewage management guidelines developed in 1992 require receiving water studies for larger treatment facilities, which can involve total P modelling and limits. Educational materials have also been developed to inform the public of potential nutrient related impacts from residential development and to promote volunteer preventative actions and stewardship of the water resource.

Watershed management plans are promoted for public drinking water supplies and in watersheds designated under the Canadian Heritage Rivers program. Various contaminants including nutrients would be addressed in both situations.

Recent policy development has prohibited commercial aquaculture operations within freshwater lakes and ponds. Land-based aquaculture operations are subject to controls limiting loading of nutrients.

The Nova Scotia Department of Environment (NSDOE) supports several site-specific water quality monitoring programs aimed at addressing local concerns regarding nutrients, including the Metro Areas Lakes Program, which monitors nutrients and phytoplankton biomass in protected areas; local community groups, which monitor nutrients and other parameters in certain water bodies; and a municipal based Volunteer Lake Monitoring Program, which helps protect lakes against the impact and extent of residential development.

For the agricultural community, there are recommended guidelines for operations (e.g., Brenton and Mellish 1996; Robinson and Gordon 1996) as well as the recently developed Green Farm Initiative. The purpose of this initiative is to use BMPs to remediate as well as to prevent environmental impacts to water resources. In an industrial context, the NSDOE has introduced a Pollution Prevention Program governing substance release.

Ontario

Direction for the management of Ontario's surface and ground waters is provided in the 1994 *Water Management Policies, Guidelines; Provincial Water Quality Objectives of the Ministry of Environment and Energy* (OMEE 1994*)*. The document provides a basis for determining "limits of the uses of the resources" that will serve to protect the province's water resources. Methods of achieving these defined limits are to be decided locally based on existing conditions and incorporating other pollution management strategies.

Provincial Water Quality Objectives (PWQOs) consist of ambient water quality criteria set so as to maintain the quality of surface waters of Ontario (Table 6.3). The criteria are for surface waters however, they may also apply to ground water where it is discharged to surface waters. Although the criteria apply to nearly all waters in the province, interim PWQOs were developed to address specific water quality concerns. For instance, recreational development of lakes often results in phytoplankton blooms and other undesirable aesthetic and biological effects. Low flow and agricultural inputs can also cause high P concentrations in watersheds. Interim PWQOs are set using total P as a general guideline for use in conjunction with site-specific studies. These interim objectives are designed to protect against aesthetic deterioration; eliminate excessive plant growth in rivers and streams; avoid nuisance concentrations of algae in lakes; and protect the diversity of trophic status while still enabling development to occur. Thus, Ontario currently has a P interim PWQO for lakes on the Precambrian Shield that allows spring overturn values to rise to 50% above predevelopment levels. The *Rationale for a Revised Phosphorus Criterion For Precambrian Shield Lakes in Ontario* (January 1998; OME 1998) cites "preventing undesirable changes in the aquatic ecosystem due to increased algal growth" as the rationale for developing new P objectives.

In terms of assessment, the Provincial Water Quality Monitoring Network measures P concentrations in receiving waters. The data are used at the regional level for receiving water assessments and in the issuing of Certificates of Approval. These Certificates are legal instruments that may limit concentrations or loadings of P. Concern over eutrophication of the Great Lakes in the 1960s, as well as obligations under the *Canada-U.S. Great Lakes Water Quality Agreement* (through the 1971 *Canada-Ontario Agreement*), led to the implementation of these measures.

Ontario has a considerable number of voluntary initiatives in place. Within the Ministry of Environment, one of the most significant initiatives is total P management. The initiative is currently being formalized and, when adopted, will control the total amount of P discharged. For example, if a discharger wishes to increase its P loading then they will usually require a deviation when the PWQO is exceeded. A deviation is acquired by providing offsets within the watershed by reducing loadings from non-point sources, such as agriculture. Some municipalities are asking for offsets of 2:1 and greater.

There are also various agricultural initiatives within the Ontario Ministry of Agriculture, Food and Rural Affairs (OMAFRA) and organizations such as the Ontario Farm Environment Coalition, AGCARE, and Ontario Soil and Crop Improvement. Ontario provides information on practices and systems related to the storage, handling and disposal of livestock wastes (including manure) in its *Ontario's Nutrient Management Strategy* (Ontario Farm Environment Coalition 1998). Ontario produces *Field Crop Recommendations* that provides farmers with technical information on nutrients and fertilizer application in order to promote good practices (e.g., OMAFRA 1994). The Ontario Farm Environment Coalition (1998) is developing an agricultural nutrient management policy and is encouraging OMAFRA to integrate this policy and associated processes into Bill 146, the proposed *Farming and Food Production Protection Act*.

Prince Edward Island

Studies conducted in PEI have identified problems of eutrophication largely caused by land management activities, notably excessive plant production in the upper portions of several estuaries. A 1997 Roundtable on Resources and Sustainable Land Use examined the impact of various PEI land uses on water resources and recommended a number of changes to the current land uses. In response to these recommendations, PEI is currently developing legislation on buffer zones to protect all water bodies situated in areas of livestock or crop production, particularly potato production.

Documented evidence of water quality problems in PEI (rising nitrate levels in surface waters) and the recent development of the proposed buffer zone legislation have served to educate the public and prompted voluntary initiatives. The agricultural community is working to improve land management through such initiatives. Special studies are often conducted to assess the impact of nutrients. In addition, PEI has developed *Guidelines for Manure Management for Prince Edward Island* that recommend manure management practices as well as provide direction for the establishment and operation of new livestock confinement facilities, expansion of existing livestock facilities, or changes in land use in rural areas (PEI Agriculture 2000).

Québec

Nutrient guidelines have been developed in the Province of Québec for the protection of aquatic life and the protection of water quality for recreational and aesthetic uses (Québec's Surface Water Quality Criteria [*Critères de qualité de l'eau de surface au Québec* (1998)]. These criteria are also used for development of receiving water objectives based on usual decontamination technologies. Modified from Ontario's objectives, Québec's provincial water quality criteria for total P are designed to protect against aesthetic deterioration; eliminate excessive plant growth in rivers and streams; avoid nuisance concentrations of algae in lakes; and protect the diversity of trophic status while still enabling development (Table 6.3).

Recently, Québec modified its criteria for total P in lakes. The new guideline allows values in lakes, during the ice-free period, to rise to 50% above natural background levels (MEFQ 1998a). As is the case for Ontario's provincial criteria, the prevention of undesirable changes in the aquatic ecosystem due to increased algal growth is the rationale behind the development of this new P objective. For protection of sensitive species habitats (i.e., lake trout), criteria must be validated with the lake ecosystem capacity method, which is based on hypolimnetic oxygen concentrations. There are also surface water quality criteria in Québec for total ammonia and nitrite, which have been adopted from criteria used in British Columbia, Ontario, and the United States (Table 6.3).

Environmental problems caused by toxic substance release from urban, industrial or agricultural sectors to water were addressed in the late 1970s by the water clean-up program, Programme d'assainissement des eaux du Québec. This Program generally regulated point sources, and was thus unlikely to address nutrient sources such as agricultural runoff. Because certain streams were impacted by nutrients (particularly P and, in certain seasons, ammonia) as a result of intense agricultural activities, regulations were developed in 1997 for the reduction of pollution from agricultural activities (*Règlementation sur la réduction de la pollution d'origine agricole*; MEFQ 1998b). This regulation provides farmers with technical plans on nutrients and fertilizer application to promote good practices. It is also designed to prevent eutrophication of water resources due to agricultural seepage and surface runoff. There are also directives for manure storage for the protection of ground water and streams, and legislation on water buffer zones that indirectly protects surface waters against nutrient releases. The legislation is unique in that, unlike other provincial manure management plans, it includes guidelines for manure application based upon soil and crop requirements for P, rather than N.

Saskatchewan

Concerns about excessive growth of aquatic weeds and algae in the Qu'Appelle river system prompted one of the first assessments of eutrophication in Saskatchewan. In 1970, the Governments of Canada, Saskatchewan and Manitoba signed an agreement to create a comprehensive framework to develop and manage the water and related land resources of the basin. To follow through on recommendations put forward under this agreement, the Governments of Canada and Saskatchewan signed the 10-year Qu'Appelle Agreement in 1975 to protect and improve the Qu'Appelle Valley's environment, resources and cultural heritage, and to promote economic growth in the area through tourism and recreation. As part of this agreement, Regina upgraded its sewage treatment facilities in 1976 to tertiary treatment, Moose Jaw diverted its sewage to agriculture through spray irrigation, and livestock operations in the basin were assessed and, if need be, relocated or upgraded.

Saskatchewan's 1995 *Surface Water Quality Objectives* aid in determining nutrient limits for point source discharges. Other controls, such as the *Water Pollution Control and Waterworks Regulations* (1989), do not address nutrients directly but provide the authority to develop site-specific requirements. Eutrophication of water bodies in the region also prompted the Riparian Habitat Initiative. Saskatchewan also has a manual outlining appropriate methods for handling, utilizing, and disposing of manure (SAF 1997).

Currently, the Ecosystem Management Special Projects Unit in the provincial Department of Environment and Resource Management is in the initial stages of determining ecological monitoring parameters that will address various water quality issues in the province. At this point, nutrients are one of the proposed indicators, though no criteria have been developed to date. The intent is to consider ecosystem impacts and therefore to look at the effects of nutrients not only on primary production but also on other elements of the aquatic food web.

Yukon Territory

Yukon aquatic and terrestrial systems are typically low in anthropogenic nutrients due to a small population and abundant water resources. Sewage from most communities in the Yukon receives at least secondary treatment, is retained in storage lagoons and then discharged to land or wetlands. Primary treated domestic sewage from the communities of Dawson and Carmacks is discharged directly into the Yukon River but effects cannot be detected largely because of the river's high discharge and cold temperatures. A large freshwater aquaculture facility has caused eutrophication of a stream, indicating that additional nutrient controls are required to maintain other water uses.

6.2. Selected Regulations, Guidelines, and Best Management Practices in Other Countries

Nutrients are managed or are of concern to many countries around the world. In the USA, 45% of all surveyed lakes, reservoirs and ponds and 30% of all surveyed river reaches are impaired by nutrients (USEPA 1998b). In England, 78 of 102 lentic sites were found to be eutrophic or hypereutrophic based on total P criteria (Carvalho and Moss 1995) and 2 540 km of 62 rivers and canals have been designated as eutrophic (UK Environment Agency 1998). A preliminary review of eutrophication in

Europe suggested the problem was pervasive, being most severe in Belgium, Denmark, Germany, Italy and The Netherlands, and moderate in Greece, Ireland, Luxembourg and Spain (Chiaundani and Premazzi 1988). The continuing concern about widespread deterioration in environmental quality due to nutrients has resulted in considerable effort directed at managing nutrients in the environment.

This section highlights some of the actions being undertaken by other countries with respect to regulations, guidelines, objectives, and best management practices to curb nutrient losses and their environmental impacts. There are basically two complementary approaches to tackling water pollution:

- ambient quality objectives that define the minimum quality requirements of ambient water, air or soil to prevent deleterious effects to aquatic life, human health, livestock, etc.
- source control that focuses on minimizing quantities of pollutants discharged from a particular source into the environment.

This section identifies some of the approaches used elsewhere to establish ambient quality objectives or to minimize nutrient losses from point and non-point sources.

Setting Ambient Quality Objectives for Nutrients

Several countries (such as the USA, Australia and New Zealand) are currently developing approaches for setting scientifically-defensible numeric nutrient and algal criteria applicable to their ecozones. A directly prescriptive approach to nutrient criteria development is not appropriate due to inherent regional differences in water quality.

In the USA, the Environment Protection Agency announced in June 1998 that it is going to require all states to develop and adopt numeric nutrient and algal criteria for all waters (USEPA 1998a). Frameworks for establishing science-based nutrient and algal criteria for rivers/streams, lakes, wetlands, and estuaries have been drafted and are currently being revised. These draft documents recommend an approach for setting criteria. For example, in the case of the rivers/streams document, criteria development would be based upon analysis of nutrient and algal data from reference and impact sites to determine conservative statistical thresholds at minimal or low impact sites and to establish nutrient-algal relationships.

Australia and New Zealand are also currently revising the Australian and New Zealand Environment and Conservation Council (ANZECC) *Water Quality Guidelines for Fresh and Marine Waters*, a document published in 1992. The main objective of the new guidelines document is to provide an authoritative guide for setting water quality objectives that will sustain current, or likely future, environmental values for natural and semi-natural water resources in Australia and New Zealand. Similar to the US EPA draft documents, the draft ANZECC document provides an approach for setting nutrient guidelines recognizing the need to tailor guidelines to specific ecosystem types (e.g., lowland rivers, upland rivers, reservoirs/lakes, wetlands, estuaries, coastal/marine waters), for specific localities or regions, and to relate nutrient concentrations to the risk of adverse effects.

In some countries (e.g., within the European Union), development of nutrient guidelines is set within the context of watershed management plans. A watershed management plan sets out a common vision for a watershed by identifying use-related objectives for water quality, along with actions to achieve the objectives. Historically, efforts at improving water quality were typically directed at managing a particular source or sector. This approach often elicited success because, prior to intensive agricultural, industrial and urban development and advances in waste treatment technology, pollution could often be tracked back to a single source. However, now, with the gains made in our knowledge of environmental science and technology, it is rare that a single source determines the nutrient status of a waterbody. The additive effects of many individual sources of nutrient loading (no one of which may be a major polluter) must now be addressed and this can only be done through watershed management plans.

A watershed management plan involves:

- assessing the impact of human activities on surface waters and ground waters in each river basin, taking into consideration pollution from point sources, pollution from diffuse sources, water abstraction and other human activities that affect water status;
- setting ecological objectives for water quality and quantity based on an assessment of water needs, and impacts of human activities on water bodies;
- exploring management options for mitigating pollution;
- establishing and implementing a program of measures to achieve the ecological objectives which involves interested parties such as relevant government departments, local communities, and water utilities;
- establishing, measuring and monitoring systems for evaluating progress towards ecological objectives; and
- reviewing effectiveness of the action program at regular intervals.

Development of watershed management plans may range from a somewhat informal arrangement between interested parties (e.g., landowners, environmental groups, and local government) to formal regulations or directives. An example of the latter is the European Commission's proposed *Water Framework Directive* which, if and when adopted, will require management by river basin. The proposed Directive will require that a river basin management plan be established for all rivers to prevent deterioration of ecological quality and pollution of surface waters; prevent deterioration of ground water quality; restore polluted surface and ground waters; and ensure a balance between abstraction and recharge of ground water (EC 1999b). The restoration measures will comprise those required by existing Directives (such as those relating to urban waste water and agricultural nitrates), supplemented by whatever additional measures are needed to control pollution sources, point or diffuse, to achieve the stated objective.

Reducing Nutrient Loading from Point Sources

Historically, point sources (industrial and municipal wastewater) were the major sources of nutrient loading to aquatic ecosystems. However, because point sources are easily identified and regulated, nutrient contributions from point sources have declined over the past four decades in most industrialized countries. Despite these successes, population growth and the demand for industrial products have resulted in point source inputs still being a significant source of nutrients to many ecosystems. Solutions to further reduce nutrient loading from point sources are available and some are described below.

Compulsory treatment of all municipal wastewater: In the European Union and the USA, treatment of all municipal sewage is or soon will be mandatory. For example, in the USA, the *Clean Water Act* includes a deadline for cities to achieve secondary treatment of sewage. This deadline was originally mid-1977. The US congress later extended that deadline to mid-1988. According to the US EPA, 87% of all communities complied with the 1988 deadline; 80% of those that missed the deadline were small cities (defined as those with less than 10 000 population). Cities that missed the 1988 deadline are now under judicial or administrative orders to comply as soon as possible.

In the case of the European Union, the European Commission's *Urban Waste Water Directive* requires all agglomerations above 2000 population equivalents (where 1 population equivalent equates to a five-day biochemical oxygen demand of 60 g of oxygen per day) to have sewer systems and secondary (i.e., biological wastewater) treatment. For certain marine waters, primary (i.e., mechanical treatment) might be sufficient provided it can be proved that water quality is not adversely affected. The deadlines for reaching the objectives depend on the size of the agglomeration and the character of the receiving water and range from 1998 (for communities > 10 000 that discharge to sensitive areas) to 2005 (for communities < 10 000). By 1998, any discharge of sewage sludge to water bodies is prohibited. The Directive also states that by December 2000, industrial wastewater from most food and beverage industries shall before discharge respect the established conditions for 4000 population equivalent or more (EC 1999a).

Advanced P removal from wastewater: Advanced P removal is a well-established technology for treating wastewater. Directives are in place in some jurisdictions requiring MWTPs discharging to sensitive water to upgrade to advanced P removal. For example, the European Commission's *Urban Waste Water Directive* requires agglomerations discharging wastewater to so-called sensitive areas (i.e., water bodies that are eutrophic or in danger of becoming so) to have wastewater treatment that is more advanced than secondary treatment (EC 1999a).

Upgrades to sewers: Repair and replacement of sewage systems have reduced pollutant loading from many cities, both in Canada and worldwide. Repair or replacement of sewers ensures minimal infiltration during wet weather and reduces leakage. Conversions of combined sewer systems to separate systems or shunting of the most toxic first flush of stormwater into storage facilities/ponds for subsequent treatment at a MWTP prevent untreated sewage from entering surface waters.

Reducing Nutrient Losses from Non-Point Sources

Large-scale agricultural production without some nutrient losses to ground and surface waters and to the atmosphere is virtually impossible. However, the impact of agriculture on water quality has become a point of concern in many nations. For example, the US EPA ranks agricultural activities as the most significant cause of impaired water quality in streams and lakes (USEPA 1994a,b); this trend is not surprising, given that agriculture is one of the most extensive users of the nation's land base (USEPA 1998b). The focus of much of this concern has been animal husbandry, where the geographic intensification of animal production in some regions has raised serious questions about the ability of the agricultural land base to assimilate animal manure nutrients without environmentally significant losses in erosion, runoff, leaching, or volatilization. Science-based solutions that can reduce agricultural nutrient losses further are available today and are described below. Improvements in environmental quality can be expected as nutrient losses are reduced.

Comprehensive Nutrient Management Plans for Individual Farms: Most provinces in Canada have in place or are developing nutrient management strategies (or "nutrient bookkeeping") for managing the production, storage and utilization of agricultural nutrients on farms. These are a valuable means to improve a farmer's ability to manage nutrients more effectively, with the ultimate aim to reduce over-fertilization. Other jurisdictions (such as several US states) also require nutrient management plans, in particular submission of manure management plans by managers of large livestock operations (usually > 300 to 1 000 livestock units [or 300 000 to 1 000 000 lbs. of livestock], depending upon the state) to the regional authority.

Regional Nutrient Management Plans for Agriculture: In areas of intensive livestock production where the available agricultural land base is not sufficient for economic and environmentally-safe application of manure, animal and crop producers can work collaboratively to transport and apply animal manure on farmland that is nutrient deficient. This can be accomplished by means of a cohesive network of farm operators that brokers the exchange of surplus manure from animal producers to crop farms so as to reduce the amount applied to concentrated areas and redistribute it over a broader agricultural base. Such a centralized network was adopted in The Netherlands where a manure board was created to ensure application of manure to impoverished soils. Here, the government is pursuing a target of equilibrium manuring such that in about 10 years time, the amount of phosphate and nitrogen applied in the form of animal manure and chemical fertilizer must be broadly equal to the amount of phosphate and nitrogen utilized by the crops.

To limit over-application of manure, it may be necessary in some cases to reorganize livestock into smaller operations so that land application is more economically feasible. For example, in The Netherlands, the government is targeting large-scale farm operations so farmers will guarantee they have contracted enough land to receive their livestock manure safely. If they cannot achieve this guarantee, they must scale back their operation. In other words, The Netherlands is trying to create a land-based livestock sector.

Other considerations for effective farm nutrient management include assessments as to the siting of intensive livestock operations or crops (e.g., corn) and the proportion of land set aside compared to land intensively cropped. For example, the distribution of arable land in Europe has changed, especially since 1993, when the Common Agricultural Policy (CAP) reform went into effect. Although the primary goal of the policy was to take arable land out of cultivation to avoid build-up of a grain, it also served to reduce the land base to which agricultural chemicals were applied.

Guidelines for Manure Application to Soils: Land application of manure and commercial fertilizers should be based on a balance between the requirements of the crops and the supply to the crops from the soil and from fertilization. These guidelines are typically based on the N application rates because N is usually the nutrient that limits crop growth. However, manure is rich in P relative to N compared to plant growth requirements. This attribute means that application of manure to achieve a desirable level of N will result in an over-abundance of P in the soil. Although most soils have a high binding capacity for P, eventually excess P will reach surface waters where it may cause eutrophication. Certain countries where manure management from intensive livestock operations has long been an issue have adopted P guidelines for manure application (e.g., The Netherlands and Belgium).

Managing Nutrient Losses from Aquaculture Operations: Countries such as New Zealand, Japan, USA and Norway have a long history of aquaculture. Regulations and/or codes of practice have been established to minimize environmental impacts, including those associated with nutrient loss. These include:

- Siting criteria to lessen impacts from nutrient losses from cages and from waste discharges and to determine if a waterbody can support an aquaculture operation.
- Open water cage technology so that cages can be placed away from restricted waters and shorelines.
- Criteria for collection and treatment of wastewater, particularly for cage operations in freshwater.
- Good management practices with respect to general site operations, waste disposal and garbage removal, and fish feeding practices.

6.3. New Technologies for Reducing Nutrient Loading

Modern engineering and environmental sciences are advancing at a remarkable pace, allowing for innovations that will increase the efficiency of nutrient use in a variety of sectors. Improvements in environmental quality can certainly be expected as nutrient losses are reduced through use of new, as well as well-established, technologies. Examples of some of the new or emerging technologies are:

Recovery and recycling of phosphate from wastewater: The chemistry of phosphate recovery and recycling appears relatively straightforward. Although industrial application of this process in municipal waste treatment facilities is still at an early stage, a number of research or demonstration installations are already running. The technology appears to hold considerable promise for recovering P, as phosphates, from MWTPs that can then be recycled back into the detergent phosphate industry and into other high-grade industrial uses. In addition, recycling of phosphates from wastewaters will reduce sewage sludge volumes and ash production where sludges are incinerated, and reduce chemicals used in sewage treatment works.

Increased nutrient retention by livestock: Livestock incorporate only 20 to 40% of the P and N originally present in the feed. Phytic acid is the main form of P in plants. Phytase (for breaking down phytic acid) is lacking in hogs and chickens (cows have phytase). Research is continuing on approaches for increasing nutrient retention by livestock and technologies are now emerging for adding phytase and/or other supplements to livestock diets (although this has to be tempered with health side-effects on animals) or to match animal diet to requirements. Plant genetic approaches are also looking at decreasing phytic acid content of plants.

Improved Feed Formulations for Aquaculture: Only 20 to 30% of the N and P added to aquaculture operations are incorporated into fish biomass and removed at harvest; the other 70 to 80% of added nutrients are lost to the environment as metabolic waste, faeces and uneaten food fragments. Improvements in feed quality have already reduced the feed coefficient (wet weight of feed used to wet weight of fish produced). New formulations should continue to focus on development of more nutritionally balanced and digestible feed to reduce waste discharges from feeding.

Treatment of Animal Wastes: With livestock and poultry industries becoming larger and more intensive, significant amounts of manure are generated that must be collected, stored, and utilized efficiently. Broad spectra of integrated manure management systems are available to collect, transfer, store, treat and efficiently utilize a great variety of sources and nutrient qualities of animal manures. Although treatment of animal waste is not a new technology, it has not been widely used as livestock and poultry producers have been given little incentive to implement such environmental practices and technologies. In areas of intensive livestock production, treatment of animal waste could be evaluated as an option for reducing the risk of manure contamination of surface and ground waters.

Nutrient Management Best Management Practices (BMPs): Agriculture's contribution to non-point source pollution varies widely as a complex function of land use, cropping system, soil type, climate, topography, hydrology, and agricultural nutrient management practices. Despite this complexity, research-based nutrient management practices that are effective at reducing non-point source pollution are available throughout many countries, including Canada. New BMPs directed at specific crops or livestock or specific geographic conditions are continually developed. Wider implementation of new, as well as currently recommended, nutrient management BMPs by agriculture is essential if significant improvements in environmental quality are to be gained.

Although BMPs are most often discussed with respect to agricultural practices, the need for BMPs also applies to other sectors involved in land management, for example land clearing for oil, gas and mining operations, timber harvest, and urban development. Although less attention has been directed at developing nutrient management practices for sectors other than agriculture, there is a growing research base on BMPs that are effective at reducing non-point source pollution for other land-based sectors. Further research and implementation of nutrient management BMPs by land management operations are essential if significant improvements in environmental quality are to be gained.

6.4. Conclusions

In Canada, the problems of excessive nutrient loading were first recognized in aquatic ecosystems, notably Lakes Erie and Ontario, because of the highly visible problems associated with eutrophication: blue-green algal scums; unpleasant odours and tastes; and unsightly beaches. Scientific assessments of the problem identified nutrients, particularly P, as the pollution culprit. Federal and provincial/territorial governments responded with a wide range of measures including legislation, regulations, objectives, guidelines, and best management practices (Table 6.1). Over the years, these have been refined as new information became available. In many jurisdictions, there has been a shift from “end of pipe” responses to a more preventative or proactive approach. This shift is a result of a growing tendency towards a broad ecosystem approach to environmental protection.

Many of the approaches used to minimize nutrient loading and associated environmental impacts in Canada are also employed in other jurisdictions. Permit limits, for example, are a well established and widely used tool for regulating wastewater discharges from industrial and municipal sewage treatment plants. However, compared to Canada, countries with a longer history of settlement and intensive agriculture have tended to adopt tighter restrictions to minimize nutrient loading to the environment. Several countries (e.g., USA, Australia, New Zealand, and countries in the EU) are also developing frameworks for setting scientifically-defensible regional nutrient criteria.

In addition to the well-established approaches for reducing nutrient impacts to the environment, new technologies continue to be developed. They range from methodologies for source control of nutrients to operational practices to reduce impacts of nutrients on the environment. Environmental policy should continue to emphasize integrating the best and most advanced science into practical solutions.

7.0 Emerging Issues

Highlights

- Phosphorus loads in Lake Erie are now about half the peak loads of the early 1970s. These reductions, in combination with the introduction of the zebra mussel, *Dreissena sp.*, have improved water clarity in Lake Erie. However, concerns about deleterious changes to the amount and type of fish harvest have been raised.

- Urea was the chemical most commonly used at Canadian airports to de-ice runways and planes. Because its primary breakdown product is ammonia, which can be toxic to aquatic organisms, airports are now switching to potassium acetate and sodium formate which are less toxic although less effective as de-icing agents.

- In the summer, a phenomenon called "white haze" occurs in southern British Columbia when ammonia from manure volatilizes into the atmosphere and reacts with urban smog particles.

- Fire retardants are essentially fertilizers designed to enhance green plant growth in order to stop the spread of the fire.

- Livestock production has intensified in many parts of Canada. Overall, the number of farms with cows, pigs or poultry decreased by 59% between 1976 and 1996. In contrast the number of cows, pigs or poultry increased by 39%.

- The production of fin and shellfish by Canadian aquaculture operations is expected to double between 1997 and 2005.

In recent years, a number of specific issues related to nutrient levels and their environmental, human-health, or economic consequences have arisen. These range from concerns that reductions in phosphorus loading to the Great Lakes may have impaired the fishery, to the effects of aircraft and runway de-icers and fire retardants on aquatic ecosystem health, to economically-viable and environmentally-safe use of sewage biosolids and livestock manure. This chapter explores the scientific underpinnings and management considerations for these emerging issues.

7.1. Low Phosphorus Concentrations in the Lower Great Lakes

The *Canada-U.S. Great Lakes Water Quality Agreement* (GLWQA) signed in 1972 contained provisions to reduce obnoxious algae problems by reductions in the loading of P. Target P loads were achieved around 1986 mainly by construction of new MWTPs and the adoption of phosphorus precipitation techniques in older MWTPs. The target loads are about one half the peak loads of the early 1970s and lakewide P concentrations have responded proportionately. Recently, the appropriateness of the GLWQA loading targets has been questioned in light of the changing lake ecology brought about by zebra and quagga mussels (*Dreissena sp.*) that invaded the lower lakes beginning in 1988. Fluctuations and declines of some fish species have been noted by the commercial and sport fishery industries. Declines of some fish species are thought to be caused by excessive grazing pressure of the mussels at the bottom of the food chain (i.e. grazing on algae). At the same time, food webs are changing with the emergence of a mussel-goby-bass food chain. Unprecedented low P values have occurred in the east and central basins of Lake Erie since the *Dreissena* invasion. These concentrations were lowest in 1995 and increased in 1996 and 1997. Although water clarity has changed little offshore in Lake Erie after the mussel invasion (Charlton et al. 1998), there has been a noticeable increase in clarity in shallower nearshore areas (Howell et al. 1996) due to the mussels' filtration (grazing) effects. Phytoplankton abundance (measured as chlorophyll concentrations) do not seem to have changed much in the offshore waters of the central and east basins coincident with the *Dreissena* invasion compared to the decrease that occurred earlier due to nutrient reduction. This shift means that although lake productivity is lower than in the early 1970s, the majority of production decrease occurred before mussels arrived. In addition, there may be delayed effects of the achievement of the nutrient load targets that reinforce, or are even more important than, the effects of mussels. The mussels can cause pronounced depletion of algae in nearshore areas (Graham et al. 1996), and this may be where the early life stages of fish are most susceptible to grazing-induced lack of food. At this time, if and how mussels affect fish populations is the subject of ongoing research.

Another issue raised regarding P in the Lower Great Lakes is whether fish populations are at risk from low P. When the P minima occur offshore in summer in Lake Erie, concentrations of P are still higher than those in Lake Huron and this reflects the nutrient load and morphometry of Lake Erie. Lake Huron still has a fishery. Thus, there is probably no threat to the survival of fish in Lake Erie from present P and chlorophyll concentrations alone. The size and type of a sustainable fish harvest may be affected, however.

One solution for perceived low fish production suggested by some fishing interests is relaxation of P removal during winter months at MWTPs. The historical data indicate that nutrient loading reduction caused a decrease in total P of around 3 µg/L in the central and east basins of Lake Erie (Charlton et al. 1998). Therefore, assuming the lake functions as before, a return to nutrient load pollution levels of the 1960s would be needed to increase P by a similar amount. Much of the P load reduction, however, resulted from control of raw sewage and partially treated sewage. Only a portion of the difference between pre-and-post P control loads would be made available, therefore, through any simple shift in technology such as cessation of P precipitation at MWTPs. Most of the sewage P load now under control, which might be made available, is discharged in the west basin. Increasing west basin loads would have little effect on P in the central and east basins unless a return to gross pollution year-round was allowed. Concerns with this idea would be the operational practicality at sewage plants, effects on landowners, drinking water quality, beach quality, and the co-emission of toxic chemicals that may otherwise be removed along with P. In addition, more nutrient loading may have other deleterious effects by stimulating production of blue-green algae that now seem to be favoured in the west basin. Of course, the conditions of tributaries and the effects of more nutrients on shoreline properties already beset by noxious accumulations of *Cladophora sp.* algae would have to be taken into account if an attempt were made to fertilize the nearshore areas or even the lake as a whole. Fishing was apparently adequate in the period 1985 to 1990 when offshore phytoplankton abundance was almost as low as in recent years. Thus, it is doubtful that whole basin nutrient additions from sewage would be sufficient to influence fish production significantly. Scientific consensus is that a change in Great Lakes phosphorus management policy is not advisable at this time.

The present sewage loading limits in the Great Lakes are regulated on an effluent concentration basis. Thus, the load of P can gradually increase due to population growth. Fortunately, more diligent application of present P removal techniques can be applied in order to maintain or lower loads from sewage. Amelioration of local eutrophication problems in "areas of concern" (identified in the GLWQA) would cause a nutrient load reduction to the lake. Non-point sources are numerically very important now and these will decline if objectives to ameliorate conditions in tributaries and to prevent loss of valuable soil are successful. Non-point P loads are mainly poorly available to algae because the P is bound in soils except immediately following rain when fields have just been fertilized. Thus, although the total P load to the lake may decrease further due to non-point controls, only a small part of that decrease would potentially affect lake productivity. The GLWQA sought to limit productivity of Lake Erie as a way of improving water quality, beach quality and fish habitat while preventing auto-fertilization of the lake from sediment phosphorus regeneration. Some decline in lake productivity would be expected but at this time we do not know whether this will be ameliorated or exacerbated by the food web shift to benthic populations.

7.2. Runway De-icers: The concern over the use of urea at Canadian airports

Every winter, major airports in the Northern Hemisphere are confronted with snow and freezing rain conditions, which play havoc with airline schedules and compromise human safety. To maintain safe conditions for travellers, airport operators apply chemicals to aircraft, taxiways and runway surfaces. The chemicals are selected for their ability to prevent formation of ice on aircraft and runways (anti-icing), or to melt ice and snow when formation has occurred (de-icing). For the past three decades, the chemical most commonly used at Canadian airports as both an anti-icer and a de-icer on runways is urea (Transport Canada 1990). Urea was selected because it does not corrode aircraft parts, stain paints, or affect the plastic of aircraft windows (Transport Canada 1994). Urea works by lowering the freezing point of water to -11.5°C. The runway temperature, the presence of winds, the amount of precipitation on the airside pavement, the chance or occurrence of rainfall and the formation of precipitation all influence application rates. Generally, in anti-icing, a spreading rate of 15 g/m^2 is sufficient (Transport Canada 1972). De-icing quantities are determined by the thickness of the ice and are applied after as much snow and ice has been removed from the runway surface as is physically possible.

The use of urea as a runway ice control chemical, although effective, poses many concerns regarding environmental contamination. Results of Canadian and European studies indicated that 64 to 100% of the urea applied at airports may be discharged directly into the environment (Transport Canada 1990). Due to the frozen nature of most soil systems during periods of urea use, the most likely receptor of this runoff is surface water. There are two primary ways in which urea can disturb aquatic life and diminish water quality: toxicity and eutrophication. Urea has a 24-h LC_{100} value (i.e., the concentration of urea that kills 100% of the individual organisms to which it is administered) of 30 000 mg/L for creek chub in freshwater (Weiss 1986). Urea is also toxic to other aquatic life and micro-organisms at concentrations above 10 000 mg/L and to domestic animals at doses above 500 mg/kg (EPS 1985).

Once in the environment, urea [$CO(NH_2)_2$; 46.7 % N by weight] degrades to ammonia, nitrite, and nitrate. The degree of contamination to any aquatic system is determined mainly by the amount of urea used at the airport, the temperature of receiving waters and soils, and the pH of receiving waters and soils. High temperatures and high pH environments are more conducive to quick degradation of urea and its N components than are cold temperatures and low pH environments (Transport Canada 1990). A 1990 study by Transport Canada to determine whether application of urea on Canadian airport runways contributed to lake eutrophication, oxygen demand, aquatic toxicity, drinking water contamination or soil contamination found unacceptable levels of the breakdown products of urea (nitrate, ammonia, nitrate) at all of the studied airports. Since then, Canadian airports have

Table 7.1. Urea usage at selected Canadian airports (tonnes). Data source: 1974-1989 data from Transport Canada 1990; 1997-1999 unpublished airport data.

Airport	**Average annual use**	
	1974-1989 (t)	**1997-1999 (t)**
Calgary, AB	NR	16
Charlottetown, PEI	60	30
Edmonton, AB	45	No longer used
Fredericton, NB	NR	55
Gander, NF	NR	475
Halifax, NS	250	250
Montreal, QC (Dorval)	450	600
Montreal, QC (Mirabel)	900	600
Ottawa, ON	100	No longer used
Quebec City, QC	NR	200
Regina, SK	NR	No longer used
Saskatoon, SK	40	5
St. John, NB	NR	60
St. John's, NF	200	225
Thunder Bay, ON	NR	No longer used
Toronto, ON (Pearson)	300	No longer used
Victoria, BC	NR	20
Whitehorse, YK	NR	2
Windsor, ON	NR	2
Winnipeg, MB	75	No longer used
Yellowknife, NWT	NR	15

NR= Not reported

implemented a program of sampling and analyzing stormwater from open channels to ensure that airport effluent does not negatively affect the environment (Transport Canada 1994; DND et al. 1998).

The toxicity and eutrophication potential of urea has spurred a search for ways to mitigate possible deleterious effects. A number of methods for lowering urea's impact on the environment have been proposed and considered including blowing of snow over a larger area to increase the uptake of N by plants; selective harvesting of airside grass to maximize the amount of N removed from the area; construction of retention walls to slow down the escape of runoff and increase infiltration into the soil; ponding of urea-laden storm and melt water for summer watering of grass; use of a vacuum sweeper to collect stormwater for treatment; and treatment of urea stormwater by various methods. However, Transport Canada (1990) deemed all of these methods to be insufficient or impractical for controlling the environmental impacts of urea use at Canadian airports. Low toxicity alternatives have also been pursued. Three such products are calcium magnesium acetate, potassium acetate, and sodium formate. From the results of initial environmental studies (Transport Canada 1994), potassium acetate and sodium formate seem to be the most promising for use as runway anti/de -icers in Canada. Although urea remains operationally superior to all other de-icers, its degradation to ammonia creates serious environmental impacts. As a result, it has been banned at airports in several countries (Transport Canada 1994).

Urea use for each airport varies from year to year depending on the climatic conditions for each de-icing season. Urea is purchased by weight at the beginning of and during the de-icing season of October to May. Urea application records for individual de-icing events are not kept. Average urea use for a number of Canadian airports (1974-1979 and 1997-1999) are given in Table 7.1. In 1996, Lester B. Pearson International Airport was the only commercial airport that used exclusively non-urea runway de-icers (e.g., potassium acetate and sodium formate) (DND et al. 1998). However, a survey in August 1999 showed that airports in Edmonton, Ottawa, Regina, Thunder Bay, and Winnipeg have joined Pearson and switched to using sodium formate as a runway de-icer and potassium acetate as a runway anti-icer. Meanwhile, the Calgary Airport, the John G. Diefenbaker Airport in Saskatoon and the Whitehorse Airport now use mostly sodium formate as a runway de-icer mixed in with a little urea. Many of the airports still using urea indicated that they have performed assessments on the use of sodium formate and potassium acetate; however, cost of the replacements has deterred them from completely switching from urea. As for Canadian Forces Bases, only one Canadian base (Moose Jaw, SK) still uses urea as a runway de-icer; the others have switched to sodium formate (Lewis Cocks, DND, personal communication).

7.3. White haze over the southern region of the Lower Fraser Valley (adapted from Environment Canada 1999d)

For years, residents in rural regions of British Columbia's Lower Fraser River Valley were puzzled by the appearance of a thick band of white haze in the sky on calm, sunny days. Recent studies identified it as a rural version of urban smog associated with intensive agricultural production and, specifically, emissions from poultry and other livestock manures.

The haze, which occurs a couple of hundred metres above the ground, can persist for days trapped by a cap of warm air that seals off the valley like a lid and obscures the view of the surrounding mountains (Figure 7.1). Although its health effects are not yet fully known, the direct relationship between fine particles, respiratory disease and mortality has fuelled growing concern over this unusual phenomenon.

Manure has been a major concern in the Fraser Valley for some time because of its connection with rising N levels in local water supplies. Nitrogen budgets constructed for the Lower Fraser Valley revealed that a large quantity of N was entering the air as ammonia. Ground-level air sampling showed high concentrations of ammonium sulfate and ammonium nitrate in the air --evidence that the ammonia was likely combining with N and sulphur oxides from industrial pollutants and vehicle emissions. When the moisture in these compounds evaporates, the fine particulates that remain are visible as a milky haze. Some of these particles fall to the earth or are washed out in rain, so they can re-enter the cycle again and again.

Results from computer modelling suggest air parcels move from Vancouver through the Fraser Valley. As these air parcels pass over the Abbotsford area, an area of intense poultry farming, they pick up high levels of ammonia. White haze has not yet been reported elsewhere in Canada, although it is also a common phenomenon in farming regions of Colorado. Current efforts to transport manure out of the area to nitrogen poor regions of the province will help reduce agricultural ammonia emissions in this area. As well, local research efforts have investigated a change in poultry diets to reduce the N content of manure, amending turkey litter with alum to reduce ammonia emissions, or using it as a nutrient additive in cattle feed. A field study planned for the late summer of 2001 will attempt to

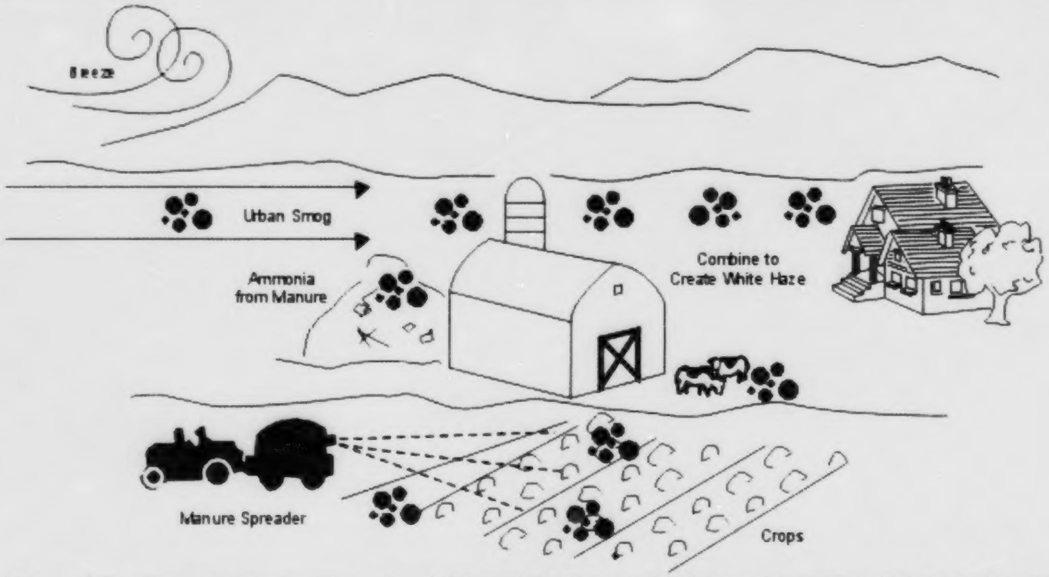

Figure 7.1. Ammonia from manure volatilizes into the atmosphere and reacts with particles of urban smog to create white haze.

measure the white haze in the Lower Fraser Valley. These, as well as existing data on white haze, will be used to develop airshed management plans and encourage stricter controls on N and sulphur oxide emissions in this rapidly growing region of the country.

7.4. The effects of nutrient enrichment on biodiversity

Biodiversity refers to "the variability among living organisms from all sources, including, among others, terrestrial, marine, and other aquatic ecosystems, and the ecological complexes of which they are part" (Glowka et al. 1994). Biodiversity can be examined at three levels of biological organization: ecosystem, species, and gene. Ecosystem diversity refers to the number, variety, and extent of ecosystems either globally or within a given geographic area. It is important because genetic and species diversity tend to be greater where there is a variety of ecosystems to which species can adapt and where there has been enough time for complex ecosystems to develop. Species diversity refers to the variety of species on Earth or within a given area or ecosystem. It is essential to the production, consumption, decomposition, recycling, and other processes that take place in ecosystems. Finally, genetic diversity refers to the extent of heritable variation among members and populations of a given species. It allows species to adapt to changes in their environment (Government of Canada 1996). Earth's biodiversity is increasingly threatened by human activities that cause loss of natural ecosystems and extinction of species (e.g., Ehrlich and Ehrlich 1981; Freedman et al. 1996). Nutrient addition to native ecosystems is one of many threats to biodiversity.

Human alteration of the N cycle has accelerated losses of biological diversity especially among plants adapted to low-nitrogen soils (Vitousek et al. 1997). Limited supplies of biologically available N are typical of most natural terrestrial and marine ecosystems, and many native plant species are adapted to function best under this constraint. New supplies of N added to these ecosystems can cause a dramatic shift in the dominant species and a reduction in overall species diversity as the few plant species adapted to take advantage of high N out compete their neighbours. For example, experimental application of N fertilizers to grasslands and heathlands in Europe and North America has dramatically reduced the biodiversity of plant species, in turn potentially affecting other ecological processes (Vitousek et al. 1997). Similar phenomena are seen in freshwater ecosystems where P (rather than N) addition is usually responsible for eutrophication and loss of diversity.

The stresses from nutrient enrichment tend to drive aquatic ecosystems to less resilient, less complex, simpler states. In short, the composition of aquatic ecosystems and natural ecological processes are compromised:

- Moderate enrichment of **lakes** tends to result in increases in phytoplankton and rooted aquatic plants, coupled with changes in aquatic plant species composition. There are also increases in the abundance and changes in the composition of aquatic animals, such as invertebrates and fish. With gross enrichment, lake bottoms become smothered, resulting in reduced biodiversity of benthic organisms, as well as losses of fish species as a result of reduced survival of fish eggs due to oxygen depletion.
- In **rivers and streams**, moderate enrichment leads to increases in periphyton biomass and rooted aquatic plants, along with increases in productivity for higher trophic levels, such as insects and fish. With gross enrichment, there is reduced productivity by periphyton; reduced productivity by benthic invertebrates and fish; and a loss of some species.
- In **wetlands**, the number of species per unit area and biomass production increase under moderate nutrient loading. At higher nutrient loading, plant growth becomes limited due to competition for light (by shoots and leaves) and for space (by roots). Plant diversity also gives way to dominance by single species due to invasion by aggressive nitrogen-loving, highly tolerant, non-native vegetation. For example, in freshwater bogs, *Sphagnum* declines in abundance and is replaced with more nitrophilous mosses, whereas in fens, there are increases in tall graminoids (grasses and sedges) and decreases in subordinate plant species that are less able to compete for light. Marshes display increases in phytoplankton and emergent vegetation. In salt marshes, waters become choked with phytoplankton, blocking light penetration to the deeper vegetation. Thus, the direction of wetland succession is altered by changes in vegetative biomass and species composition. There may be shifts in invertebrate species composition, which can lead to changes in the quality and quantity of waterfowl food. Amphibians (frogs, toads, and salamanders) may be subject to nitrate and nitrite toxicity. Large amounts of nutrients released from salt marshes to adjacent estuaries may stimulate coastal water food webs.
- **Coastal waters** show evidence of increases in fast-growing phytoplankton and macro-algae, and reduced light to seagrasses, diminishing their ability to photosynthesize. Sediment anoxia impairs the ability of seagrasses to acquire N and accelerates seagrass mortality. Turbidity increases as sediments are no longer stabilized, further favouring motile phytoplankton that can move to the surface to maximize exposure to light; anoxia also enhances release of nutrients from the sediments (internal nutrient loading), favouring phytoplankton. Habitat is lost for many fish and benthic organisms. Marine algae have been found responsible for massive mortalities of fish, bird, and marine mammals.

Coupled with the stresses of increased nutrients is the fact that industrial emissions of nitrogen oxides further stress the biodiversity of species and ecosystems as they give rise to acid precipitation. Results include soil and water acidification, reductions in availability of nutrients for plants and animals for both terrestrial and aquatic ecosystems, and death of aquatic organisms (Government of Canada 1991). In conclusion, maintaining diversity helps ensure that the Earth will continue to perform the natural ecological processes upon which all life depends.

7.5. Fire retardants: Forest saviours or destroyers?

An important challenge currently facing forest managers in Canada is the development of ecosystem management plans that preserve biodiversity, allow multiple use of forested lands, and promote sustainable use. Forest fires play an important role in maintaining ecosystem health as they reset the successional stage of a forest. However, fires also endanger lives and cause economic loss to the forestry industry and property owners. Canada uses a variety of chemicals to aid in the suppression of wildfires, including long- and short-term fire retardants, foams, and wetting agents. These fire-fighting practices may have significant ecological consequences. For example, chemicals used to suppress wildfires may be applied to environmentally sensitive areas containing endangered, threatened, or economically important plant and animal species (Hamilton et al. 1994). Although it is not feasible to adopt non-suppression policies for all wildfires, concern exists about local impacts of retardants due to their potential toxicity or nutrient enrichment effects (Duchesne 1994).

Long-term fire retardants are formulations that continue to reduce or inhibit combustion after the water in the chemical-water mix has evaporated (Labat-Anderson Inc. 1996). They are usually applied aerially to a 25-30-m strip around the perimeter of a fire as the fire spreads, changes direction or approaches a town (Albert Simard, NRCan, personal communication; NARTC 1993). The retardant mixture consists of about 85% water, 10% fertilizer (typically ammonium phosphate or ammonium nitrate), and 5% minor ingredients (colour, thickener, corrosion inhibitors, stabilizers, and bactericides) (George 1995). Thus, long-term fire retardants are basically fertilizers. The fertilizers are present to encourage growth of vegetation both in the short-term, to make it more resistant to the fire, and over the long term following the fire (Johnson and Sanders 1977). The application of long-term fire retardants adjacent to and onto surface waters has resulted in four major environmental concerns: ammonia toxicity to sport fish; toxicity caused by the breakdown products from individual ingredients in the retardant; decline in species number and diversity of stream organisms; and eutrophication (Labat-Anderson Inc. 1996).

The greatest environmental threat posed by long-term fire retardants is ecosystem enrichment. Although it is preferable that aerial drops of fire retardant into or along any body of water be avoided, the fire location or the nature of the terrain may result in some retardant being dropped into lakes or streams. The possibility of eutrophication is greatest when retardant is dropped directly onto a stream or lake, particularly if the N content of the receiving water is very low. The eutrophication risk posed by long-term fire retardants to aquatic systems depends therefore upon the volume of retardant dropped onto a water body, the volume of the water body, and the flow or water exchange in the stream or lake. The risk of eutrophication caused by runoff from vegetated strips treated with fire retardant is considered to be small as most of the P is retained by the soil in a non-leachable form (Norris et al. 1978). Moreover, by the next year, most of the nutrients have been consumed by the forest vegetation (Albert Simard, NRCan, personal communication).

Long-term fire retardants usually pose a small toxicity risk in the environment. For aquatic ecosystems, the toxicity risk depends upon the ability of the receiving water to dilute the fire retardant. In general, long-term retardants are practically nontoxic to fish, with 96-h LC_{50} values ranging from 90 mg/L for Coho salmon yolk-sac fry to greater than 10 000 mg/L for rainbow trout eggs (Johnson and Sanders 1977; Poulton et al. 1993). Long-term retardants are slightly toxic to aquatic invertebrate species, with 96-h LC_{50} values for scud ranging from 45 to 62 mg/L (Johnson and Sanders 1977;

Monsanto 1991). Long-term fire retardants do not appear to pose an acute hazard to adult birds, mammals, earthworms, or humans (Smith 1987; Chemonics 1991; Monsanto 1991; Stillmeadow 1991; Poulton et al. 1993; Vyas et al. 1994). However, the fertilizer contained in the retardant may, under very specific conditions, cause nitrate poisoning to animals consuming hay or other forage crops contaminated by the retardant (George 1995; Labat-Anderson Inc. 1996). Little is known about the potential effects of these formulations on vertebrate behaviour and population dynamics. There is also little information on the potential toxicity of long-term fire retardants to terrestrial plants. It is possible that aerosols of ammonium sulfate may cause injury, including yellowing, tissue death and wilting, of plant leaves (Bradstock et al. 1987).

Currently, it appears that long-term fire retardants used to control Canadian forest fires do not pose a threat to the flora and fauna when used appropriately. However, further research is needed to address the impacts of these fertilizing chemicals when they are applied to areas that are naturally nutrient poor or nutrient rich, because the addition of the retardants could cause a dramatic shift in trophic status, even if the effect is only temporary.

7.6. Agriculture Intensification

The last century has seen great development in many agricultural technologies such as high yielding crop varieties, chemical fertilizers, pesticides, irrigation and mechanization (Matson et al. 1997). These developments have resulted in agricultural operations becoming increasingly specialized so that most farms now fall into one of two classes: (1) high density livestock operations, which often lack an adequate land base for environmentally-safe manure application, and (2) intensive cash-crop farms, which require high applications of chemical fertilizer. Where mixed farms were able to efficiently recycle animal manure by applying it to agricultural fields, the geographic separation between intensive livestock operations and cash-crop farms has resulted in poor use of manure as a fertilizer in some locales. This inequitable distribution of resources has raised concerns about the effect of P and N from manure and chemical fertilizers on air and water quality (Matson et al. 1997, Gleig and MacDonald 1998).

Manure is a low-density nutrient source and thus is not economical to transport long distances (Gleig and MacDonald 1998). Between 1976 to 1996, livestock operations in Canada became fewer and larger. For example, the number of hogs produced increased by 91% whereas the number of farms with pigs decreased by 69% over the same twenty-year period (Figure 7.2; Statistics Canada 1997c). Similarly, chicken numbers increased by 17% whereas chicken farms decreased by 72% while the number of cattle increased by 8% whereas beef farms decreased by 37% (Figure 7.2; Statistics Canada 1997c). For intensive livestock operations, manure poses a serious management problem as it must be stored for at least part of the year and it should be field-applied at rates that match the nutritive needs of the crop. In Canada, approximately 1 000 million tonnes of manure (wet weight) are produced annually (Larney 1992; Statistics Canada 1997c). Without proper storage, handling and field application of animal wastes, odour complaints, eutrophication of surface water, and ground water contamination become major concerns (Environment Canada 1998b; McCulloch et al. 1998). Mismanagement of manure may cause fish kills (see Chapter 5.2), contribute to the proliferation of toxic blue-green algal blooms, nitrate contamination of ground water, and eutrophication of surface waters.

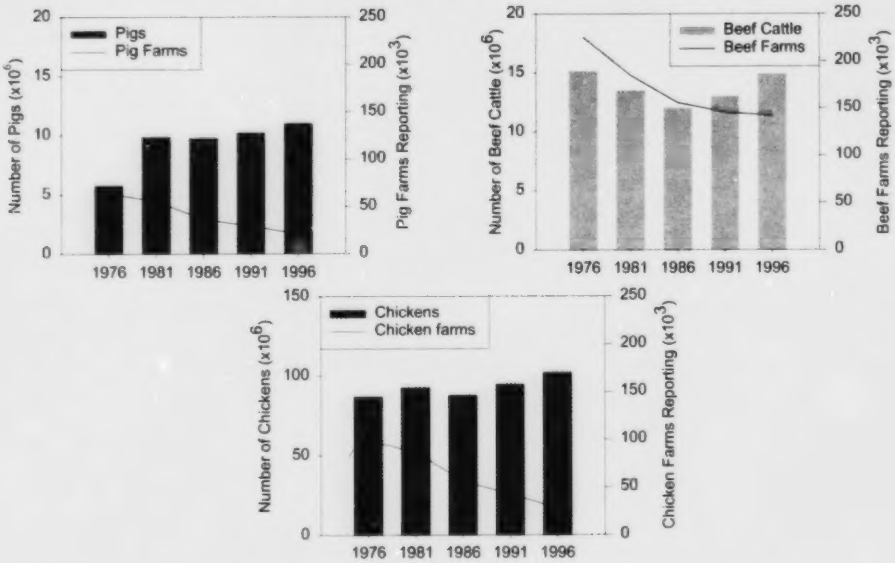

Figure 7.2. The changes in the numbers of pigs, beef cattle, and chickens and their associated farm numbers in Canada from 1976 to 1996. (Data from Statistics Canada 1997c).

In the high intensity livestock areas of Canada, centred in Ontario, Québec and Alberta, there are more nutrients available from manure than are required by crops (Bailey and Buckley 1998; Gleig and MacDonald 1998). Traditionally, manure application rates are calculated based on the N needs of the crop. This method of calculation, however, does not account for the manure's P content, which will exceed crop requirements if manure is applied on the basis of its N content. At current application rates, P levels in the soil are increasing and so is the risk of P migration and thus contamination of surface and ground water especially in the humid regions of Canada (Bailey and Buckley 1998; Gleig and MacDonald 1998). In Québec, British Columbia and Ontario, high soil P levels in areas around intense livestock production are becoming an environmental problem (Bailey and Buckley 1998).

If high intensity livestock areas have a surplus of available nutrients from manure, the opposite is true in the intensive cash cropping areas of Canada. Here, chemical fertilizers are required to meet the nutritional demands of the crop. The inequitable distribution of livestock and crops therefore limits the use of manure, not only as a fertilizer but also as a rich source of organic matter. Organic matter in the soil increases the nutrient and water holding capacity of the soil, influences soil temperature and nutrient mineralization, and provides a home for a diverse animal, bacterial and fungal community (Matson et al. 1997). The loss of the organic soil layer created by the soil biota has resulted in an increased reliance on irrigation, mechanical tillage and chemical fertilizers, all of which represent economic inputs for the producer.

Another feature of the intensification of farming is an overall reduction in plant diversity. Although managing a monoculture appears easier from a producer's perspective, this reduction in plant species creates an ecosystem favouring insect pests and plant pathogens. Nitrogen fertilization has also been found to increase the number of crop pathogens and populations of sap-feeding insects such as aphids, leafhoppers and planthoppers (Matson et al. 1997). Greater crop losses due to pests lead to an increased requirement for the use of pesticides. Recent studies have also shown that fertilizer drift promotes the growth of unwanted introduced weeds in the herbaceous layer of non-target habitats adjacent to cultivated fields (Boutin and Jobin 1998; Kleijn and Verbeek 2000), therefore only exacerbating the problem of weed invasion in cultivated fields.

Over the last 50 years, improved agricultural technology has vastly increased annual crop yields in Canada. These improvements, however, have increased the impact of agriculture on the environment, especially with respect to water quality. The agricultural community has responded with the development of best management plans that attempt to balance the continued food production needs for Canada while attempting to minimize negative effects to the environment. In Canada, manure storage management issues have been addressed through regulations, guidelines, and codes of practice under existing legislation (e.g., British Columbia, Saskatchewan, Alberta, and Manitoba) and through assistance to producers to improve manure storage and management (e.g., Québec) (Government of Canada 1996). Similarly, guidelines to help producers calculate manure and fertilizer application rates based on crop requirements and existing soil conditions determined through soil testing are produced by each province in an effort to avoid problems associated with over or under fertilizing crops. New technologies have also alleviated some manure management problems. For example, techniques such as liquid manure injection systems that incorporate manure into the soil have reduced nutrient losses. Composting of animal manure has resulted in a more stable product not as susceptible to nutrient loss and environmental contamination. Advances in biotechnology are also increasing nutrient retention by animals and thus reducing wastage.

7.7. Municipal Biosolids Disposal

Sewage sludge is a nutrient rich, organic by-product of municipal wastewater treatment. Its creation during wastewater treatment is unavoidable and as effluent quality objectives continue to increase, so does the amount of sludge produced. Biosolids are the portion of the sewage sludge that has been stabilized through digestion to meet public health regulations for application to land (OMEE & OMAFRA 1996, WEF 1998).

Biosolids are highly valued as a soil amendment and nutrient source for agriculture (Campbell and Webber 1997; Evans 1998; Mackenzie 1998). However, because biosolids are derived from household and industrial waste, concern exists that application of biosolids to arable land will result in accumulation of heavy metals or organic contaminants in the soil and, ultimately, in crops or livestock. Biosolids vary widely in their heavy metal and organic contaminant content, depending on the mix of industries contributing waste to a MWTP. At present, the concentration of heavy metals in biosolids is the primary factor used to determine land application rates (CBCL Ltd. 1996; OMEE & OMAFRA 1996; AEP 1997; Campbell and Webber 1997; BCELP 1998a; SERM 1998). Organic contaminants found in biosolids include chemicals such as polynuclear aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs), phthalate esters, dioxins, and furans. Identifying and treating the mixture of organic chemicals in sludge is a difficult, time consuming and expensive task, and thus organic contaminant content has not been used to determine land application rates. Organic contaminants, however, were not found to accumulate in Ontario agricultural soils treated with biosolids at rates greatly exceeding recommended values (Webber 2000). The introduction and enforcement of industrial surcharge programs aimed at controlling pollution at the source and the recognition of the economic benefits of waste stream recovery and reuse have resulted in a considerable decrease in heavy metal concentrations in biosolids during the 1980s (Campbell and Webber 1997). This trend suggests that as technology continues to better detect and treat contaminants in sewage sludge, contaminant concentrations will continue to decline.

Currently, the volume of biosolids produced in Canada is increasing as more Canadians are connected to sewers and sewage treatment. In addition, higher levels of wastewater treatment mean greater biosolids production. Limited storage space for biosolids and increasing tipping fees at landfills have necessitated new management strategies for biosolids. Until recently, incineration was the preferred treatment technology as it minimized the quantity of material to be disposed. However, public concern about air emissions and ash disposal has caused this management solution to fall out of favour even though the technology required to clean air emissions is available, albeit expensive (Campbell and Webber 1997). Land application is currently the preferred management option for biosolids as it is cost effective, especially for smaller operations, and there is little environmental risk if application guidelines are carefully followed. Composting and fertilizer production also take advantage of the nutrients and organic compounds in the biosolids while reducing the volume of material produced. Composting operations, however, often fail due to odour complaints and a lack of market for the product. In fertilizer production, the biosolids are dried, or pellitized, and combined with other materials to be marketed as an organic fertilizer (Campbell and Webber 1997). Other emerging management technologies for biosolids include thermal conversion into liquid fuel (oil) for use as a recoverable energy source, utilization of biosolids ash in building materials, and technologies such as aqueous phase oxidation, alkaline stabilization and chemical or biological stabilization to immobilize or eliminate toxic compounds (Campbell and Webber 1997).

7.8. Aquaculture Expansion

Over the last 20 years aquaculture in Canada has developed into an industry representing about 1 750 producers and a wholesale value of $338 million, accounting for approximately 1.5% of total global aquaculture production in 1997. Although Canada is a small player in the global aquaculture industry, Canada's small to medium operations are important sources of employment to the largely rural areas in which they are situated. Since 1991, price declines, a moratorium on new sites in British Columbia, and an outbreak of an infectious disease in New Brunswick salmon have reduced the projected rapid expansion in the Canadian aquaculture sector. Aquacultural production is predicted, however, to double in Canada between 1998 and 2005 (DFO 1999). Such growth must occur within a consistent framework of sound environmental management practices designed to maintain a healthy and productive aquatic environment.

Aquaculture operations interact with the environment in a multifaceted manner. Although a number of concerns have been voiced about potentially deleterious impacts arising from aquaculture operations, from a water quality standpoint, the greatest impact results from the addition of food pellets to the cages. Food formulations that maximize N and P conversion into fish growth and thus minimize food waste to the environment are serving to decrease this impact. Nevertheless, in situations, such as Lac Heney, Québec, where water circulation is insufficient to dilute nutrient inputs adequately, nutrient loading from aquaculture can permanently alter the food web (see Lac Heney case study, Chapter 4.2). As well, oxygen depletion due to the microbial breakdown of accumulated food waste under aquaculture cages is often responsible for altering the benthic invertebrate community. In the case of shellfish aquaculture in Canada, the concern with respect to nutrients is not losses to the surrounding water, as shellfish operations in Canada are not supplied with additional food. Rather, increasing anthropogenic nutrient loads in marine environments appear to be increasing the magnitude and occurrence of hazardous algal blooms. A hazardous algal bloom in the vicinity of a shellfish aquaculture operation necessitates its closure as filter-feeding shellfish accumulate the toxins in their flesh making them unfit for human consumption.

In October 1999, British Columbia lifted its four-year moratorium on aquaculture expansion, an action expected to be followed by a large growth in Canadian aquaculture (BCEAO 1997). Still fraught with opposition from environmental groups, the expansion of aquaculture is being touted as a viable alternative to the wild salmon fishery, creating jobs for unemployed fishers. However, aquaculture operations, if not run properly, have the potential to affect biological and physical environments adversely. Potential effects will vary widely and depend on the physical setting and scale of the operation, site maintenance, feeding and the extent of competing uses for the environment in the area. The challenge, then, is to ensure that proper management measures are in place as Canada's aquaculture operations expand.

7.9. Municipal Wastewater Treatment Plants Siting

Effluents from MWTPs are often discharged to rivers and lakes in concentrations that can be damaging. Effluents are a concern because several of their constituents such as P and ammonia are many times more concentrated than is desired in the ambient environment. For example, in Lake Ontario the ambient desired P concentration is 0.010 mg/L but MWTP discharge concentrations are typically 50 to 100 times higher (0.5 to 1.0 mg/L). Thus, the receiving water is used to dilute the effluent, resulting in a zone of dilution near the discharge location. Dilution of effluents in receiving waters is an important factor in minimizing the costs of wastewater treatment. However, even if the total quantity of a particular pollutant that is discharged is acceptable, the size of the dilution zone and the concentration of the pollutant and environmental degradation in the dilution zone may still be issues. Size and local effects of the dilution zone depend on volume and concentration of the effluent, the efficacy of the diffuser (if one is used), the dispersion characteristics (water movement and turbulence) of the receiving water, and location of the discharge.

For rivers, the choice of a discharge site depends on environmental as well as economic factors. The location, upstream versus downstream as well as the distance across the river, may be chosen to minimize local effects. Discharges close to riverbanks may hug the riverbank and proceed many kilometres downstream with only gradual mixing or dilution. Discharges in the middle of rivers can mix across the flow in two directions and thus can be diluted somewhat closer to the discharge point. Discharge diffusers can cause rapid mixing of the effluent in ambient water with associated concentration decrease.

In lakes, the length of the discharge pipe is also an economic and environmental consideration. Lengthier pipes cost more but offer less environmental damage (assuming the total amount of constituents such as P is appropriate). When mixing zones impinge on shallow water or shorelines, the high nutrient concentrations in treated sewage can cause algal blooms and accumulation of noxious attached algae such as *Cladophora*. The longer the pipe the deeper the water in which an effluent can be discharged. The deeper the water, the more lake water moves past the discharge site to dilute the effluent. Buoyant plumes can be entrained into more water when the site is deeper. Thus, the extent of high concentrations in mixing zones in surface water is minimized with longer and deeper pipes. In addition, discharges far offshore minimize impacts on water intakes. Again the further offshore the discharge, the less likelihood there is of shoreline effects.

Volumes of discharges are gradually rising as population grows. Thus, increasing volumes of effluent are being discharged in the same locations as were established decades ago. Most discharges in large lakes are within 1 to 2 km of shore. As the effluent volume increases, so does the size of the mixing zone. The impact of the mixing zone can be managed somewhat with more and more advanced sewage treatment. Eventually, however, there will be a trade off between more and more costly treatment versus more and more costly pipelines that extend further offshore. In North America, even very well treated sewage is extremely active in stimulating algal growth. The natural dispersion and assimilation abilities of lakes depend on depth and distance offshore. Thus, as volumes of sewage grow, more consideration will have to be given to discharge location if deleterious nearshore effects are to be avoided.

7.10. Lake Winnipeg: Eutrophication of a Manitoban Great Lake

Lake Winnipeg is located in central Manitoba. It is the 10th largest (23,750 km^2) freshwater lake in the world and its drainage basin is the second largest in Canada. The basin spans four provinces (Alberta, Saskatchewan, Manitoba and Ontario) and three US states, and includes three major rivers (Red, Winnipeg and Saskatchewan). Lake Winnipeg is the central lake in a chain of large lakes extending across Canada from Great Bear Lake and Great Slave Lake in the Northwest Territories to Lake Athabasca situated on the Alberta/Saskatchewan border, to Lake Manitoba and then to the Great Lakes on the Ontario-USA border.

Lake Winnipeg has considerable economic, aesthetic and cultural value to Manitobans. The landed value of the pickerel and whitefish fishery exceeds $25 million annually and supports more than 700 fishers and their communities, many of whom are aboriginal. Lake Winnipeg also serves as a reservoir for Manitoba Hydro and contributes to the generation of export power, valued in excess of $350 million annually. Lake Winnipeg is also the source of drinking water for many aboriginal communities located along the Nelson River.

The tributaries draining into Lake Winnipeg are naturally variable in their nutrient and silt load, reflecting both the parent geology and human activity in their basins. Tributaries arising to the east of the lake drain mostly the boreal shield ecozone and are characterized by high water yields and low nutrient and silt content. In contrast, waters from the south and west drain sedimentary parent material and are characterized by low water yields and high nutrient and silt concentrations.

Changes in human activity in the basin over the past 30 years have increased stress on the Lake Winnipeg ecosystem. It appears there has been a gradual increase in N and P likely due to changes in agricultural practices, including expansion of the livestock and food processing sectors, and increases in human population. Urban populations have continued to grow with several major cities still having no P removal from wastewaters. The lake has also been affected by increased demand and value for hydro-electric generation, which has led to hydraulic management of Lake Winnipeg as a reservoir. This management action may be changing the lake's nutrient carrying capacity through increased light penetration and the retention and recycling of N and P (M. Stainton, DFO, Winnipeg).

Only a few studies have been conducted on Lake Winnipeg. Bajkov (1934) in the 1920s and Brunskill (1973; Brunskill et al. 1980) in 1969 undertook the only comprehensive studies dealing with spatial and temporal variability in the physical, chemical and biotic components of the lake. Five smaller studies have been undertaken since 1969 by staff from Fisheries and Oceans Canada and the Province of Manitoba (Manitoba Department of Mines, Resources and Environmental Management 1974; Todd et al. 1996; Stewart et al. 1998; M. Stainton, DFO/Winnipeg, unpublished data). Together, these data indicate that increased human activity in the basin has caused Lake Winnipeg to become more eutrophic. Although the effects of the variety of human activities operating in the Lake Winnipeg basin are not fully understood, the deterioration in water quality is likely caused by a combination of factors. These include increased P loading from the watershed, increased transparency of the lake water due to reduced sediment input from tributaries dammed for hydroelectric power generation, and increased retention and recycling of nutrients in the lake due to changes to hydraulic cycles stemming from lake impoundment. A much better understanding of changes in nutrient loading from the watershed to the lake and changes to internal nutrient processing is required to manage inputs and effect clean-up of this great lake.

8.0 Conclusions

8.1. Issue

Nitrogen (N) and phosphorus (P) are natural resources for which there is intense competition in terrestrial and aquatic ecosystems not greatly affected by human activity. Until recently, the supply of N and P for most plants, and ultimately to animals, was limited. The most abundant N source, N gas, could only be used by plants once it was fixed by certain bacteria or algae into ammonium or nitrate compounds. Similarly, the most abundant P source, P-bearing minerals, only became available by weathering. Consequently, N or P was a limiting nutrient in most ecosystems prior to human settlement and agricultural development. Moreover, because N and P were in high demand, these nutrients were stored and recycled in close proximity to the locale from which they were scavenged. This pattern was true for plants and animals, including humans because, prior to urbanization, faeces from livestock and humans along with other composted waste was returned to the soil, thereby closing nutrient recycling loops and maintaining the fertility of the soil.

The amount of N and P available for plant uptake has increased dramatically in the past several decades. The causes are a massive increase in the use of fertilizer, burning of fossil fuels, development of large urban populations, and an upsurge in land clearing and deforestation. The amount of available N has more than doubled since the 1940s, with human activities contributing 210 million tonnes per year to the global supply of N, compared to only 140 million tonnes generated per year by natural processes (Vitousek et al. 1997). Similarly, natural weathering of phosphate-bearing rocks is now overshadowed by mining activities as a source of P, with approximately 140 million tonnes of phosphate-bearing rock now mined each year.

This influx of nutrients has disrupted the natural cycles of both N and P. Where animal manure and human wastes were historically spread on farmland to recycle nutrients, a “once-through” system now predominates. Thus, phosphates extracted from mined phosphate rock and inorganic N fixed from N gas by industrial processes are applied to agricultural land or fed to livestock. Nutrients in the form of foodstuffs flow from the farm to the cities, where most ultimately end up in landfill (sewage sludge, incinerator ash), or in surface or ground waters. Reactive N and P released to the atmosphere as a result of agricultural practices, as industrial emissions and, in the case of N, as by-products of home heating and automobile engine combustion are transported and deposited hundreds or thousands of kilometres from their origin.

This once-through system of nutrient use is not economically sustainable. In the case of P, the known reserves of currently exploitable phosphate rock are about 3.6 to 8 billion tonnes P_2O_5 with another 11 to 22 billion tonnes P_2O_5 considered potentially available if the necessary investments were made (Steen 1998). At the current rate of consumption (31 million tonnes P_2O_5 in 1995-1996 and allowing for intensification of agriculture in developing countries), current economically exploitable reserves could be exhausted in 100 years (Steen 1998). Moreover, high-grade phosphate reserves are already depleted from some sources and the quality of phosphate rock used worldwide is declining (i.e., phosphate rock reserves are now lower in their P content and higher in their contaminant content, specifically heavy metals, iron and aluminum) (Steen 1998).

In addition to the economic consequences of the breakdown in nutrient recycling, the environmental consequences resulting from addition of bioavailable N and P to the Earth's ecosystems could be profound. Globally, one of the best-documented problems caused by N overload is the eutrophication of estuaries and coastal seas (Nixon 1996). Eutrophication in coastal waters has led to increased anoxia and hypoxia, loss of seagrasses, alteration of foodwebs, loss of biological diversity, and increased incidences of nuisance algal blooms (Howarth 1993). A recent report from the US National Academy of Sciences concluded that "perhaps the most pressing problem in many estuarine and marine systems today is that of nutrient enrichment" (Boland et al. 1993). Similarly, additions of anthropogenically-derived P have increased eutrophication of aquatic ecosystems throughout much of the world (Caraco 1993).

8.2. Impacts of Nutrients on Canadian Environments

There is clear evidence that nutrients are causing problems in certain Canadian ecosystems. The severity and geographic extent of nutrient-induced environmental problems are not yet as great as in countries with a longer history of settlement or agricultural production (e.g., many European countries). Our assessment of the effects of anthropogenic nutrients on Canadian ecosystems showed that aquatic ecosystems (lakes, rivers, freshwater and coastal wetlands, and coastal waters) are subjected to some of the highest nutrient loading and show the greatest nutrient impairment. The addition of nutrients to inland waters typically exhibits a two-phase response whereby moderate nutrient addition to relatively low-nutrients waters stimulates aquatic plant and animal productivity and increases biodiversity (Table 4.6). However, prolonged nutrient addition results in excessive aquatic plant growth, loss of plant species (particularly rare species), depletion of oxygen, and deleterious changes in abundance and diversity of aquatic invertebrates, fish and possibly birds and mammals dependent upon these habitats. Ground waters across Canada also exhibit elevated nitrate concentrations that have human health implications for consumers of this water and may contribute to the eutrophication of surface waters (Table 4.6). Although concentrations of N are also elevated in the atmosphere, the ecological consequences from atmospheric deposition of N are not as pervasive as those observed from loading to surface waters. Atmospheric deposition of inorganic N is greater in eastern Canada (east of the Manitoba-Ontario border) and there is evidence that some N limited forests have responded with increased productivity and that some forested watersheds in central Ontario and southern Québec are in the early stages of N saturation (Table 4.6).

Based upon our review of available scientific evidence, we are certain that N and P loading from human activity has:

- accelerated eutrophication of certain rivers, lakes and wetlands in Canada, resulting in loss of habitat, changes in biodiversity and, in some cases, loss of recreational potential (Figure 4.10; Table 4.6).
- increased the frequency and spatial extent to which the drinking-water guideline for nitrate has been exceeded in ground waters across Canada and caused economic burden for those Canadians who must transport household water from off-site sources (Table 5.10).

- caused and continues to cause fish kills in Southwestern Ontario due to ammonia toxicity (Table 5.7; Table 5.8; 'Agricultural Water Pollution: Fish Kills in Southwestern Ontario' text box, Chapter 5.2).
- contributed to the decline in amphibians in southern Ontario due to long-term exposure to elevated nitrate concentrations (Amphibians in Danger' text box, Chapter 5.2).
- led to elevated risks to human health through increased frequency and spatial extent of toxic algal blooms in Canadian lakes and coastal waters ('Contaminated Mussels' text box, Chapter 5.2).
- contributed to acidification of soils and lakes in southern Ontario and Québec and resulted in incipient N saturation in some forested watersheds (Chapter 4.1).
- increased carbon production in Canada's forests due to N deposition (Chapter 4.2). Uncertainty remains, however, as to whether the productivity gains resulting from increased N deposition along with a warming climate and elevated CO_2 levels in the atmosphere have offset the carbon loss caused by timber harvest, fire and insect-induced mortality.
- increased concentrations of the potent greenhouse gas N_2O and increased concentrations of N oxides that contribute to the formation of photochemical smog in certain Canadian cities (Chapter 2.1).
- contributed to quality of life concerns for Canadians through water use impairments (e.g., excessive algal and aquatic weed growth; Table 4.6); aesthetic (taste and odour) concerns related to water supplies (e.g., due to geosmin and 2-methylisoborneol) (Chapter 4.2); and contamination of water supplies (e.g., by nitrate and by trihalomethanes (THMs) produced as by-product of disinfection of water containing organic material) (Chapter 5.3).
- increased the economic burden to Canadians as a result of the need for treatment, monitoring and remediation of contaminated water (Chapter 6.1).

8.3. Sources of Nutrients to Canadian Surface and Ground Waters

Municipal Wastewater

Municipal wastewater is the largest point source of N and P to the Canadian environment (Table 3.20). In 1996, an estimated 6 thousand tonnes of total P and 80 thousand tonnes of total N were released to lakes, rivers and coastal waters from Canadian households. This loading occurred despite the fact that 73% of Canadians were served by municipal sewer systems and at least 94% of the wastewater collected by sewers received primary or higher treatment.

Phosphorus discharges from municipal wastewater treatment plants (MWTPs) have decreased while N discharges have increased in recent years. Phosphorus discharge has decreased by 37% since 1983 and by 20% since 1991 as compared to 1996. This trend is due to the implementation of advanced P removal at many MWTPs discharging to inland waters. In contrast, most municipal wastewater discharged to coastal waters is either untreated or receives only primary treatment. Consequently, P loading to Atlantic or Pacific coastal waters has remained constant or increased in areas with increasing population. Nitrogen removal is rarely employed at Canadian MWTPs because of the high infrastructure costs associated with N removal technology and because most fresh waters are limited by P. Nitrogen loading to Canadian surface waters has therefore increased in response to population increases, and in 1996 was 17% higher than in 1983 and 7% higher than in 1991.

In addition to routine discharge of wastewater, MWTPs may occasionally experience a discharge of raw sewage (known as a bypass). In most provinces, these are not allowed except in emergency situations, such as protecting basements from flooding, preventing damage to MWTP equipment and averting the wash-out of solids. Data from Ontario in 1991 showed that 37% of MWTPs reported a bypass and the volume released during all bypasses equalled 0.6% of the total effluent volume treated. Bypasses of even small volumes are important because the P concentration in raw sewage is about ten times that in treated sewage.

Although most Canadians are served by MWTPs, approximately 25% are served by septic disposal systems. These systems were originally designed for houses widely separated from their nearest neighbour; however, today, septic disposal systems are present in many parts of the country in numbers too great for their land base, for example in the Lower Fraser Valley. In addition, older septic systems are not always properly maintained and some were built in unsuitable sites. In general, septic systems retain about 20 to 55% of the N and 25 to 40% of the P entering the system, with additional P retained in the drain field (Ryding and Rast 1989). Based on a 1996 population, an estimated 1.9 thousand tonnes of P and 15 thousand tonnes of N were released from septic systems in Canada. The released N and P can move into ground water and, from there, to surface waters.

Most of the N and P in household sewage is from human waste. In the case of N, more than 90% of the household load is derived from urine and faeces. P sources are more varied than for N. Analysis of 1996 data showed that human waste was the largest contributor to municipal P loading in Canada followed by commercial and industrial sources.

Discharges from Municipal Sewers

In urban areas, snowmelt or stormwater runoff empties into the municipal sewer system. Sewer systems built before the early 1940s convey both raw sewage and stormwater (so-called "combined sewers"). When rain or snowmelt exceed the capacity of the sewer system or MWTP, raw sewage and stormwater are released from outfalls into rivers, lakes or coastal waters, an event known as a combined sewer overflow (CSO). Urban areas developed since the 1950s typically have two sewer systems: one that collects raw sewage (i.e., sanitary sewers) and a second system that conveys stormwater (i.e., storm sewers). The quantity and quality of both combined sewer overflow and stormwater vary temporally and regionally, and are not routinely measured. In general, combined sewer overflows represent about 5% of the national MWTP annual discharge; they are typically more dilute than raw sewage but stronger in concentration than stormwater or treated sewage. Stormwater volume is comparable in volume, however, to the 4 300 million cubic metres discharged annually by MWTPs, although it is discharged only during wet weather. The majority of the N and P in combined sewer overflows and stormwater is in a particulate form and, thus, does not contribute immediately to eutrophication. The most immediate consequences of stormwater release or combined sewer overflows are bank erosion and elevated water turbidity and bacterial counts.

Industrial Wastewater

Most light industries discharge their wastewater into the municipal sewage system for treatment at the MWTP; therefore they are not necessarily required to report on the quality of their wastewater. However, some municipalities have sewer use bylaws that regulate the strength of industrial discharges to sewer systems. MWTPs are designed to reduce bacterial contamination and biological oxygen demand, remove most P and, in some cases, convert ammonia to nitrate. There is little ability to do more than volatilize or dilute many industrial wastes.

Industries not connected to a municipal sewage system must obtain operating permits from the government of the province in which they are situated or, in the case of industries operating in the Territories, by the federal government (unless this responsibility has been devolved to the Territory). This permit may stipulate the quantity and/or quality of waste that can be disposed.

Because not all provinces provided industrial release data and because not all industries that release wastewater are required to measure nutrients, comparisons of N and P loads among regions and among sectors cannot be made. At least 2 048 tonnes per year of total P, 7 588 tonnes per year of nitrate-N (NO_3-N), and 4 231 tonnes per year of ammonium-N (NH_4^+-N) are discharged from permitted industries to surface waters in Canada (Figure 3.6).

Agricultural Runoff and Leaching

Nutrient balances for Canada in 1996 indicated that crop removal of N and P was 89% and 87%, respectively, of nutrients added as manure, fertilizer and, in the case of N, atmospheric deposition and legume fixation. The portion remaining in the field (i.e., the difference between input and output) represents a nutrient surplus. Expressed on an areal basis, the 1996 N surplus for Canada was 4.3 kg/ha for the total area of agricultural land (68 million ha) or 8.4 kg/ha for cropland (35 million ha). Similarly, P surplus for Canada was 0.8 kg/ha for total area of agricultural land or 1.6 kg/ha for cropland in Canada. It should be noted, however, that these are average values and N surplus or deficits may occur at a local or regional scale. For example, a recent analysis of residual N in Canadian farmland reported values of $\geq$ 41 kg/ha in the lower Fraser Valley of British Columbia, the corridor of agricultural land from Lethbridge through Red Deer to Edmonton in Alberta; the Melfort area in northeastern Saskatchewan; the Red River Valley in Manitoba; southwestern Ontario, the area around Lake Simcoe and the lower Ottawa Valley in Ontario; the St. Lawrence Lowlands in Québec and the region south of Québec City; the Annapolis Valley in Nova Scotia; and the St. John River Valley in New Brunswick (MacDonald 2000a).

Of the surplus N or P measured in a field, only a portion actually moves from the field to ground or adjacent surface waters. Although national estimates are not available for nutrient losses from agricultural fields to surface and ground waters, a recent assessment of N losses from agricultural land where the soils have a water surplus predicted 17% of Ontario, 6% of Québec and 3% of Atlantic farmland would produce runoff or seepage water with > 14 mg N/L. (Macdonald 2000b). In British Columbia, 5% of the agricultural land has a water surplus and 69% of this area was predicted to generate runoff or seepage water with N concentrations > 14 mg/L. Information on P losses from agricultural fields in Canada is not yet available.

Other sources of agricultural nutrients to surface waters are discharges such as milk house wastes that drain through connected tile drains to streams, cattle allowed unlimited access to streams, erosion of stream banks due to livestock trampling or tillage, and greenhouse waste. Feedlots, in particular, are an unprecedented concentration of nutrients in the form of imported food and resultant waste into a small area. There are few regional or no national data on these types of losses.

Releases associated with Aquaculture Operations and Fisheries Enhancement

Nutrient release from fish production systems ("aquaculture") results from excretion of dissolved or solid waste and from unconsumed feed. In addition to the number of fish cultivated, the quantity and quality of the feed are the most important factors that determine the generation of nutrient release in the form of feed wastage and excretion (Persson 1991; Cho and Bureau 1997). Emissions and treatment of effluents from tank or pond aquaculture are controlled by provincial legislation; cage finfish aquaculture occurring directly in water bodies is of more concern. Unlike terrestrial feedlots and agricultural operations that have been under pressure for decades to reduce nutrient pollution of water, cage aquaculture waste is entirely released to the surrounding aquatic ecosystem.

In 1996, 53 thousand tonnes of finfish were harvested from aquaculture operations in ten Canadian provinces. Of this production, 58% was in marine environments and 42% in freshwater operations, the majority occurring in British Columbia and New Brunswick. Phosphorus and nitrogen losses for the total Canadian finfish production in 1996 were 204 t/yr P and 956 t/yr N to inland waters, and 282 t/yr P and 1 320 t/yr N to coastal waters. By comparison, a population of 100 000 served by secondary sewage treatment releases 25 t/yr P and 365 t/yr N as wastewater.

In addition to commercial aquaculture operations, direct fertilization has been used to enhance the production of sport fish, especially anadromous sockeye salmon, in oligotrophic lakes and rivers in British Columbia. Nutrients are added in an attempt to enhance food supply for salmon fry. For example, in Kootenay Lake, British Columbia, five years of N and P additions of 47 t P/yr and 206 t N/yr resulted in increased zooplankton densities, which increased both spawner size and fecundity of Kokanee salmon (Ashley et al. 1997).

Runoff from Landfill Sites

There are no national estimates of nutrient losses from landfills, although localized inputs may be significant. For example, municipal landfills in the Fraser River Basin, British Columbia released an estimated 145 tonnes of ammonia-N per year to surface waters in 1995; an additional 6 tonnes per year of ammonia-N was leached from pulp and paper mill landfills (Gartner Lee Ltd. 1997).

Forest Management Practices

Undisturbed forested lakes and streams on uplands of the Canadian Shield generally contain low concentrations of NO_3^-, NH_4, and PO_4 (Nicolson 1988; D'Arcy and Carigan 1997; Carginan et al. 2000). The relative areal extent of peat and beaver pond is important to the amount of TP, NH_4, and TON exported from forested catchments (Dillon et al. 1991). Disturbance of forested ecosystems has had variable effects on dissolved nutrient movement in soil, and on subsequent nutrient loss to streams and lakes (Krause 1982; Feller and Kimmins 1984; Nicolson 1988; Titus et al. 1997). Forest management practices that disrupt the normally efficient cycling of nutrients between the soil and trees may produce increases in streamwater concentrations of dissolved N and, to a lesser extent, P. For example, paired watershed studies have been used to demonstrate that clearcutting of coastal coniferous forests (Feller and Kimmins 1984) or temperate hardwood forest (Krause 1982) can produce short-term increases in NO_3^- concentration in streams. Comparisons of Boreal Shield lakes in reference and harvested watersheds have shown high total P and total organic N concentrations and higher phytoplankton abundance in the disturbed lakes immediately after timber harvest (Carignan et al. 2000; Planas et al. 2000).

In general, N or P fertilizers are not routinely applied to forests in most of Canada. In British Columbia, N is occasionally applied to coastal forests. Experimental studies suggest that most of the N applied to a coastal fir forest (Preston et al. 1990) and boreal pine forests (e.g., Morrison and Foster 1977) was retained by the trees and soil, although low recoveries of added N have been reported (Preston et al. 1990). Short-lived NO_3^- increases in soil leachate (Pang and McCullough 1982) and stream water (Heatherington 1985) have been reported. The impact of N fertilizers on streams and lakes, nevertheless, should be short–term and minimal if fertilization with nitrate based fertilizers and direct application to surface waters are avoided.

Nutrient losses to stream water from forest management practices have been defined for relatively few sites in Canada. It is difficult to generalize about the effect of forest management practices on water quantity and quality, regionally or nationally, because of the large climatic, topographic, soil and vegetation diversity across the country that must be considered when assessing management impacts.

8.4. Sources of Nutrients to Canadian Soils

Fertilizer and manure nutrients applied to agricultural lands are essential to maintain crop yield and soil health. Only a small portion of these nutrients is returned in the form of agricultural (livestock and crop) production. In 1996, 1 576 thousand tonnes of N and 297 thousand tonnes of P as fertilizer were applied to cropland in Canada (Table 3.22). In addition, 384 thousand tonnes of N and 139 thousand tonnes of P were applied as manure. Biosolids (the portion of sewage sludge that has been stabilized to meet regulations for land application) may also be applied to agricultural land. Of the estimated 500 thousand tonnes of biosolids produced annually in Canada (early 1980s data; OECD 1995), approximately 42% is applied to agricultural land and supplies 8 thousand tonnes of N and 5 thousand tonnes of P.

In addition to desirable nutrient inputs to soils (e.g., commercial fertilizer, manure and biosolids), nutrients are also added as a result of waste disposal. Solid waste refers to the garbage collected from households and industries that requires incineration or disposal in a landfill. An estimated 10.5 million tonnes of residential solid waste were collected by municipalities across Canada in 1992 (Environment Canada 1996c). The majority of residential solid waste is landfilled, with the remainder incinerated, recycled or composted, in the case of some organic waste. As regards industrial solid waste, approximately 10 million tonnes were generated in Canada in 1995. Of this, 78% was landfilled and 20% was recycled, with the remainder incinerated (Statistics Canada 1998b). N and P release from the breakdown of solid waste is not known.

Nitrogen and phosphorus also enter terrestrial (as well as aquatic) ecosystems from atmospheric deposition. Nitrate and ammonia deposition is considerably higher in Eastern than in Western Canada as a result of industrial activities in Central Canada and the Northeastern USA. N deposition (as nitrate and ammonium) averages 3.4 kg N/ha/yr east of the Manitoba-Ontario border compared to 0.8 kg N/ha/yr west of the border (Table 3.18). Fewer data are available on dry deposition of N; typically dry deposition accounts for a much smaller fraction of atmospheric N loads than wet deposition. In contrast to N, P deposition is not routinely measured. The few studies conducted found that P inputs from wet and dry deposition together ranged from 0.01 to 0.74 kg/ha/yr. An estimated 43 thousand tonnes of N fall on cropland, 117 thousand tonnes of N fall on farmland not used for growing crops, 182 thousand tonnes of N fall on inland waters and 1 378 thousand tonnes of N fall on non-agricultural land in Canada as a result of atmospheric deposition of nitrate and ammonium.

8.5. Sources of Nutrients to the Canadian Atmosphere

N is released to the atmosphere largely as nitrous oxide (N_2O), other N oxides (NO_x) and ammonia (NH_3).

Nitrous oxide (N_2O) is largely produced from the combustion of coal, petroleum products and natural gas, and from the use of manure or commercial N fertilizers. In 1995, 60 thousand tonnes of N_2O-N were released to the atmosphere from Canadian non-agricultural sources. Most (31 thousand tonnes) of this N was released as a result of the combustion of fuel for transportation (i.e., in car, truck, airline and boat engines) (Table 3.21). Only 3 thousand tonnes of N_2O-N were due to industrial combustion-related emissions. Another 24 tonnes of N_2O-N were emitted to the atmosphere as a result of manufacturing nitric and adipic acid. Nitrous oxide can also be released from agricultural soils following the application of manure or inorganic N fertilizers. In 1996, 38 thousand tonnes of N_2O-N were released as a result of agricultural activities in Canada (i.e., direct emission from agricultural soils, direct emission from grazing animals and animal waste management, and indirect emission from volatilization, leaching and runoff) (Desjardins and Keng 1999).

Nitric oxide and nitrogen dioxide (i.e., NO_x) are also produced during the combustion of fossil fuels. Canadian nitrogen oxide emissions for 1995 were 750 thousand tonnes N, excluding agricultural emissions for which no data were available (Environment Canada 1999a; Table 3.21). Fossil fuel combustion attributable to transportation (i.e., cars, trucks, airlines and boats) contributed 393 thousand tonnes. Another 189 thousand tonnes of NO_x-N were emitted as a result of industrial processes. Agriculture emissions are estimated to be of similar magnitude to industrial emissions (Janzen et al. 1998).

Anthropogenic release of ammonia largely arises from the storage and handling of manure and fertilizer, adding 570 thousand tonnes of ammonia-N to the atmosphere in 1995 (Vézina 1997; Table 3.21). An additional 27 thousand tonnes of ammonia-N were released to the atmosphere by industrial emissions, most from industries engaged in the manufacture of N fertilizers

Recent data are not available for P release to the atmosphere. A 1978 assessment reported that 3 402 tonnes of P were released to the atmosphere by Canadian sources (Environment Canada 1983). Of this, an estimated 1.98 thousand tonnes were from industrial processes, most of which were associated with the production of fertilizer. Another 783 tonnes of P were lost to the atmosphere during fertilizer application. The contribution from incineration of solid waste (residential and industrial) to the atmosphere was estimated to be 123 tonnes of P.

8.6. Summary

This review has demonstrated the national scope of nutrient-related impacts on aquatic and terrestrial ecosystems in Canada. In response to the objectives identified for this assessment, namely "to determine whether or not nutrients in general are causing negative environmental effects; whether only certain nutrients, rather than nutrients as a class, are problematic, and; whether those effects are limited ... to water or to entire ecosystems, including wildlife" (Government of Canada 1995), we conclude that:

- ***nutrients released to the environment from human activity are impairing the health of certain ecosystems, contributing to quality of life concerns for Canadians and, on occasion, endangering human health.*** There is clear evidence that N and P loading from human activity has contributed to eutrophication of certain rivers, lakes and wetlands in Canada; caused fish and amphibian lethality; increased the frequency and spatial extent to which the drinking-water guideline for nitrate has been exceeded in ground waters across Canada; led to elevated risks to human health through increased frequency and spatial extent of toxic algal blooms in Canadian lakes and coastal waters; contributed to the acidification of soils and lakes in southern Ontario and Québec and resulted in incipient N saturation in some forested watersheds; increased concentrations of the potent greenhouse gas N_2O and increased concentrations of N oxides that contribute to the formation of photochemical smog in certain Canadian cities; contributed to quality of life concerns for Canadians through water use impairments and aesthetic concerns related to water supplies; and increased the economic burden to Canadians as a result of the need for treatment, monitoring and remediation of contaminated water.
- ***these nutrient impacts tend to be associated with certain nutrient forms.*** Most lakes and many rivers in Canada are intrinsically P limited. Consequently, addition of bioavailable P (i.e., P in the form of phosphate, PO_4^{3-}) has accelerated eutrophication of certain lakes, rivers, and wetlands. In contrast to inland waters, the complex nature of coastal environments, high flushing rates, and inputs of N-rich deep oceanic waters have generally minimized widespread effects of anthropogenic nutrients on these ecosystems although localized problems exist. Nitrate contamination is of concern with respect to ground water reserves used as sources of drinking water. Concerns about nutrients in the atmosphere also relate primarily to N because of its role in urban smog production and greenhouse warming. Atmospheric deposition of nitrate and ammonium has increased carbon production in Canada's forests. Atmospheric deposition of nitrate has also contributed to the acidification of soils and lakes in southern Ontario and Québec.
- ***although the predominant and most demonstrable impacts to date have occurred in aquatic ecosystems and caused water use impairments, the first symptoms of negative effects on forest ecosystems have also been observed.*** For example, atmospheric deposition of nitrate has contributed to acidification of soils and lakes in southern Ontario and Québec and resulted in incipient N saturation in some forested watersheds. Deposition of ammonia and nitrate has also increased carbon production in certain forests in Canada; however, associated changes in plant and wildlife biodiversity as seen in countries with high levels of atmospheric N loading have not been reported in Canada to date.

8.7. Information Gaps

This assessment identified the effects of anthropogenic nutrients on Canadian ecosystems and the sources of these nutrients. We have documented deleterious changes in Canadian ecosystems as a result of nutrient loading and the impacts of these changes on the quality of life of Canadians (see Chapter 8.2). However, our ability to assess changes in ecosystems in response to added nutrients was constrained by data limitations. These limitations could largely be divided into two categories: insufficient monitoring data of emissions and discharges and ambient conditions, and insufficient knowledge as to the effects of nutrient additions to ecosystem and human health.

Insufficient monitoring data: Although every attempt was made to define the status of Canadian ecosystems with respect to nutrients, data on sources and impacts became progressively less available as one moved from lakes to rivers/streams to wetlands to ground water to coastal waters to forests. Topics requiring particular attention are:

- industrial loading to surface waters. At present, the availability of N and P data for industries not connected to MWTPs is erratic: monitoring and reporting requirements vary among provinces and territories, and among industrial sectors. Of the 2 130 industries in Canada with discharge permits, we obtained data on nutrient loading for only 91 for nitrate, 142 for ammonium, and 191 for total P (Table 3.6). Moreover, the data are not stored in any single database.
- MWTP loading to surface waters. At present, data on N and P loading are available for certain MWTPs in Canada but the data are not consistent in parameters measured or frequency of sampling. In addition, the data are not stored in any single database. Our analysis of nutrient loading from MWTPs was achieved by applying per capita nutrient loading coefficients to the population served by the various levels of sewage treatment.
- agricultural loading to surface and ground waters. Although studies have been conducted at the scale of plots, fields or small watersheds, regional or national estimates of nutrient loading to surface and ground water could not be calculated.
- atmospheric deposition of P and total N. Although estimates of atmospheric deposition of inorganic N are available through a network of provincial and federal monitoring sites, similar data are not available for P or total N, nor are estimates available for release from various sectors.
- ground water quality. Water well survey programs are patchy across the country. Some wells are already above or close to guidelines for nitrate and bacteria. Little information is available on ammonia and P in ground water.
- fish kills from accidental spills/discharges of nutrient-related compounds. Currently, reporting is on a voluntary basis.

Effects of nutrient addition on ecosystem and human health: Nutrient management is a persisting environmental issue unlike others, such as toxic chemicals that can be eliminated by reformulation or discontinuance. Additional research is required to understand the effects of added nutrients on Canadian ecosystems. Areas requiring particular attention are:

- fate and transport of nutrients within different ecosystems (wetlands, coastal waters, forests, rivers, and lakes) and effects on biota.
- the role of nutrients in inducing blue-green algal blooms and toxin production.
- the role of nutrients in causing taste and odour problems in drinking-water supplies.
- interactions of nutrients with organic contaminants and their effects on aquatic food webs.
- effects of sewage and industrial wastewater plumes on aquatic life during periods of ice cover (i.e., limited mixing of the plume and cold water temperatures).
- effects of long term (decades) of nutrient loading on aquatic and terrestrial ecosystems, including water and sediment/soil quality and food webs.
- effects of forest management practices on nutrient loss from forests to aquatic ecosystems.
- cumulative effects on the aquatic environment from the combination of several nutrient sources all operating within a region.
- the relationship between nutrient concentrations and aquatic plant biomass, particularly for streams and coastal waters, and the level of nutrients that begins to impair beneficial uses of streams for the preservation of aethetics/recreation and protection of aquatic life.

8.8. Prognosis for Future

Over the past 100 years, the demand for water has increased steadily in Canada. This is partly a result of population growth and increasing municipal demands for water. It is also related to energy-intensive industrial development, expansion of irrigated agriculture, and rising living standards.

The Brundtland Commission called for sustainable development that meets the needs of this generation without compromising the ability of future generations to meet their needs. Maintaining the quality of the air, water and soil environments with respect to nutrients is an important component of sustainable development. Yet studies have already shown that in parts of the world, a surfeit of fixed N (compounds such as ammonia and nitrogen oxides) is overwhelming ecosystems ranging from forests to coastal waters. This N, largely from synthetic N fertilizers and nitrogen oxides discharged from cars and industries, can no longer be absorbed by terrestrial ecosystems and so is ending up in rivers, lakes, estuaries and oceans. This additional N is causing coastal algal blooms that can be toxic and can deplete the water's oxygen supply when they sink and decay. In addition, excess N is also adversely affecting terrestrial ecosystems by displacing ions from forest soils, thereby leading to mineral deficiencies and ultimately death of forest trees. In 1997, both the Ecological Society of America and the International Scientific Committee on Problems of the Environment named N pollution as a "pre-eminent problem" that is not being given enough public recognition.

This report has clearly demonstrated the symptoms of environmental degradation from anthropogenic nutrient loading in Canada. At present, environmental problems caused by excessive nutrients are less severe in Canada than in countries with a longer history of settlement and agricultural production. This trend is due to our relatively small population compared to our land base and the protective measures implemented by both the federal and provincial/territorial governments in the last 30 years. However, while successes have been realized, environmental and human health problems related to nutrients are evident across Canada. Unlike countries with long histories of settlement and agriculture and, consequently, more severe pollution, we are currently in the position of being able to deal with nutrient pollution before it is overwhelming. Science-based solutions are available that can assist in further reducing nutrient losses and, in turn, improving environmental quality. New technologies are emerging that can minimize nutrient loading to the environment. Options for reducing nutrient loading are available, particularly from countries with a long history of nutrient problems. Nutrient-related research and monitoring are essential to ensure decision-making is based upon sound science. It is critical that the gains achieved by improved wastewater treatment and other control measures not be reversed by relaxation of standards or by failure to keep pace with population growth, and that the best and most advanced science be integrated into practical solutions.

References

Aaby, B. 1994. Monitoring Danish raised bogs. *In* Mires and Man. Mire Conservation in a Densely Populated Country- the Swiss Experience. *Edited by* A. Grünig. Birmensdrof, Kosmos. p. 284-300.

AAFRD (Alberta Agriculture, Food, and Rural Development). 2000. Code of Practice for Responsible Livestock Development and Manure Management. <www.agric.gov.ab.ca/agdex/400/400_27-2.html> (5 Dec. 2000).

Abelovich, A. and Y. Azov. 1976. Toxicity of ammonia to algae in sewage oxidation ponds. Appl. Environ. Microbiol. 31: 801-806.

Aber, J.D., A. Magill, R. Boone, J.M Melillo, P. Steudler and R. Bowden. 1993. Plant and soil responses to chronic nitrogen additions at the Harvard Forest, Massachusetts. Ecol. Appl. 3: 156-166.

Aber, J.D., A. Magill, S.G. McNulty, R.D. Boone, K.J. Nadelhoffer, M. Downs and R. Hallett. 1995. Forest biogeochemistry and primary production altered by nitrogen saturation. Water Air Soil Pollut. 85: 1665-1670.

Aber J., W. McDowell, K. Nadelhoffer, A. Magill, G. Bernston, M. Kamakea, S. McNulty, W. Currie, L. Rustad and I. Fernandez. 1998. Nitrogen saturation in temperate forest ecosystems. Bioscience 48: 921-934.

AC [Agriculture Canada]. 1989. Backgrounder 89-01: The characterization and identification of potentially leachable pesticides and areas vulnerable to groundwater contamination by pesticides in Canada. Food Production and Inspection Branch. Ottawa, ON. 37 p. + Appendicies.

Ackefors, H. and M. Enell. 1990. Discharge of nutrients from Swedish fish farms into adjacent sea areas. Ambio 19: 28-35.

Adachi, R. 1965. Studies on a dinoflagellate *Peridinium polonicum Woloszynska*. I J. Faculty of Fisheries 6: 318-326.

Adams, M.A. 1993. Discussion paper: Purple loosestrife (*Lythrum salicaria*) in the Fraser River Estuary. Fraser River Management Program. Tech Report H-93-01. Fraser River Estuary Program Office, New Westminster, BC. 26 p.

Adaros, G., H.J. Weigel and H.J. Jäger. 1991. Single and interactive effects of low levels of O_3, SO_2, and NO_2 on the growth and yield of spring rape. Environ. Pollut. 72: 269-286.

Admiraal, W. 1977. Tolerance of estuarine benthic diatoms to high concentrations of ammonia, nitrite ion, nitrate ion, and orthophosphate. Mar. Biol. 43: 307-315.

AEP [Alberta Environmental Protection]. 1997. Guidelines for the application of municipal wastewater sludges to agricultural lands. Edmonton, AB.

Aerts, R. and F. Berendse. 1988. The effect of increased nutrient availability on vegetation dynamics in wet heathlands. Vegetatio 76: 63-69.

AES-EC [Atmospheric Environment Service, Environment Canada]. 1997. 1997 Canadian acid rain assessment volume two: Atmospheric science. Atmospheric Environment Service, Air Quality Branch, Downsview, ON.

Agriculture and Agri-Food Canada. 1997. Profile of production trends and environmental issues in Canada's agriculture and agri-food sector. Minister of Public Works and Government Services. Publication 1938/E. 46 p.

Agriculture and Agri-Food Canada. 1998. Canadian fertilizer consumption, shipment and trade. 1996-1997. Farm input market unit, Ottawa, ON.

Agriculture Canada. 1992. Ontario farm groundwater survey: Winter 1991/ 92. Ottawa, ON.

Agriculture Canada. 1993. Ontario farm groundwater quality survey, Summer 1992. Report prepared under the Federal-Provincial Land Management Assistance Program. *Edited by* D. Rudolph and M. Goss. ISBN 0-662-20879-X.

Ahl, T. 1988. Background yield of phosphorus from drainage area and atmosphere: An empirical approach. Hydrobiol. 170: 35-44.

Alabaster, J.S., D.G. Shurben and G. Knowles. 1979. The effect of dissolved oxygen and salinity on the toxicity of ammonia to smolts of salmon, *Salmo salar*. J. Fish. Biol. 15: 705-712.

Alabaster, J.S., D.G. Shurben and M.J. Mallett. 1983. The acute lethal toxicity of mixtures of cyanide and ammonia to smolts of salmon, *Salmo salar* L. at low concentrations of dissolved oxygen. J. Fish. Biol. 22(2): 215-222.

Alberta Environment. 1999. Surface water quality guidelines for use in Alberta. Environmental Science Division. Edmonton, AB.

Alexander, G.J. and R.J. Stevens. 1976. Per capita phosphorus loading from domestic sewage. Wat. Res. 10: 757-764.

Alexandrov, V.A. and W. Jänisch. 1971. Teratogenic effect of ethylurea and nitrite in rats. Experientia 27: 528-539.

Allan, R.J., J.D.H. Williams, S.R. Joshi and W.F. Warwick. 1980. Historical changes and relationship to internal loading of sediment phosphorus forms in hypertrophic prairie lakes. J. Environ. Qual. 9: 199-206.

Al-Mufti, M.M., C.L. Sydes, S.B. Furness, J.P. Grime and S.R. Band. 1977. A quantitative analysis of shoot phenology and dominance in herbaceous vegetation. J. Ecol. 65: 759-791.

Amundson, R.G. and D.C. McLean. 1982. Influence of oxides of nitrogen on crop growth and field: an overview. *In* Air Pollution by Nitrogen Oxides. *Edited by* T. Schneider and C. Grant. Elsevier, Amsterdam. p. 501-510.

Anderson, A.-M., D.O. Trew, R.D. Neilson, N.D. MacAlpine and R. Borg. 1998. Impacts of agriculture on surface water quality in Alberta. Part 1: Haynes Creek Study. Canada-Alberta Environmentally Sustainable Agriculture Agreement (CAESA).

Anderson, D.M. 1994. Red tides. Sci. Amer. 227: 62-68.

Antler, S. 1997. Composting comes of age: Highlights from a new Canada-wide study. Solid Waste and Recycling. October/November 1997. p. 12-17.

Aravena, R., M.L. Evans and J.A. Cherry. 1993. Stable isotopes of oxygen and nitrogen in source identification of nitrate from septic systems. Ground Water 31: 180-186.

Armstrong, D.C., D. Chippendale, A.W. Knight and J.E. Colt. 1978. Interaction of ionized and unionized ammonia on short-term survival and growth of prawn larvae, *Macrobrachium rosengerii*. Bio. Bull. 154: 15-31.

Arp, P.A. and U. Oja. 1992. Estimating base cation losses and forest biomass response to acid sulfate/nitrate loading: a case study for maple sites within the Turkey Lakes (Algoma) and Harp Lake (Dorset) watersheds. CEA No. 841-g-726, Joint CEA-CPPA-FC-EC Project: Tech. Rep., Part XVI.

Arp, P. A., T. Oja and M. March. 1996. Calculating critical S and N loads and current excedences for upland forests in southern Ontario, Canada. Can. J. For. Res. 26: 696-709.

Arthur. J.W., C.W. West, K.N. Allen and S.F. Hedtke. 1987. Seasonal toxicity of ammonia to five fish and nine invertebrate species. Bull. Environ. Contam. Toxicol. 38: 324-331.

Ashley, K., D. Sebastian, L. Thompson, D. Lasenby, L. McEachern and K. Smokorowski. 1997. Restoration of an interior lake ecosystem: the Kootenay Lake fertilisation experiment. Water Qual. Res. J. Can. 2: 295-323.

Atkinson, C.J., P. Wookey and T.A. Mansfield. 1991. Atmospheric pollution and the sensitivity on barley leaves to abscisic acid and carbon dioxide. New Phytol. 117: 159-166.

Bacchus, H. 1974. An ecological study of Cootes Paradise. MSc. thesis. McMaster University, Hamilton, ON. 220 p.

Bailey, L. and K. Buckley. 1998. Land Application of hog manure: Agronomic and environmental considerations. The Canadian perspective. Presentation at the HEMS Workshop, Toronto, ON. April 27-28.

Bajkov, A. 1930. Biological conditions of Manitoba lakes. Contrib. Ca, Biol. Fish. 5: 383-422

Baker, J. and V. Waights. 1993. The effect of sodium nitrate on the growth and survival of toad tadpoles (*Bufo bufo*) in the laboratory. Herpetol. J. 3: 147-148.

Baker, J.M. and V. Waights. 1994. The effects of nitrate on tadpoles of the tree frog (*Litoria caerulea*). Herpetol. J. 4: 106-108.

Balmer, P. and B. Hultman. 1988. Control of phosphorus discharges: present situation and trends. Hydrobiol. 170: 305-319.

Barica, J.M. and F.A.J. Armstrong. 1971. Contribution by snow to the nutrient budget of some small Northwest Ontario lakes. Limnol. Oceanogr. 16: 891-899.

Barr Engineering. 1993. Phosphorus reduction study for the Twin Cities Metropolitan Area. Report prepared for the Minnesota Pollution Control Agency, St. Paul, MN, USA.

Barry, D.A., J.D. Goorahoo and M.J. Goss. 1993. Estimation of nitrate concentrations in groundwater using a whole farm nitrogen budget. J. Environ. Qual. 22: 767-775.

Basu, B.K. and F. R. Pick. 1996. Factors regulating phytoplankton and zooplankton biomass in temperate rivers. Limnol. Oceanogr. 41: 1572-1577.

Bates, S.S., C.J. Bird, R.K. Boyd, A.S.W. de Freitas. M. Falk,. R.A. Foxall, L.A. Hanic. W.D. Jamieson, A.W. McCulloch, P. Odense, M.A. Quilliam, P.G. Sim, P. Thibault. J.A. Walter and J.L.C. Wright. 1988. Investigations on the source of domoic acid responsible for the outbreak of amnesic shellfish poisoning (ASP) in Eastern Prince Edward Island. Atlantic Research Lab Tech. Report 57. NRC, Halifax, NS. 59 p.

Bates, S.S., C.J. Bird, A.S.W. deFreitas, R. Foxall, M. Gilgan, L.A. Hanic, G.R. Johnson, A.W. McCulloch, P. Odense, R. Pocklington, M.A. Quilliam, P.G. Sim, J.S.C. Smith, D.V. Subba-Rao, E.C.D. Todd, J.A. Walter and J.L.C. Wright. 1989. Pennate diatom *Nitzschia pungens* as the primary sources of domoic acid, a toxin in shellfish from Eastern Prince Edward Island, Canada. Can. J. Fish. Aquat. Sci. 46: 1203-1215.

Bates, S.S., A.S.W. de Freitas, J.E. Milley, R. Pocklington, M.A. Quilliam, J.C. Smith and J. Worms. 1991. Controls on domoic acid productivity by the diatom *Nitzschia pungens* f. Multiseries in culture: Nutrients and irradiance. Can. J. Fish. Aquat. Sci. 48: 1136-1144.

Bates, S.S., J. Worms and J.C. Smith. 1993. Effects of ammonium and nitrite ion on growth and domoic acid production by *Nitzschia pungens* in batch culture. Can. J. Fish. Aquat. Sci. 50: 1248-1254.

BCAF [British Columbia Ministry of Agriculture and Food]. 1999. Environmental guidelines for producers. <www.agi.gov.bc.ca/resmgmt/fppa/pubs/environ/envguide.htm> (3 Feb. 1999).

BCEAO [British Columbia Environmental Assessment Office]. 1997. Salmon aquaculture review: Volume 3: Technical advisory team discussion papers. Queen's Printer for British Columbia, Victoria, BC.

BCELP [British Columbia Environment, Lands and Parks]. 1998a. Draft Selected Organic Matter Recycling Regulation. Victoria, BC.

BCELP [British Columbia Environment, Lands and Parks]. 1998b. Solid waste in British Columbia. <www.ene.gov.bc.ca:80/sppl/soerpt/waste/waste.html> (6 April 1998).

BCELP [British Columbia Environment, Lands and Parks]. 1999. British Columbia water quality status report. <www.elp.gov.bc.ca:80/wat/wq/public/bcwqsr/bcwqsr1.html> (4 Aug. 1999).

BCF [British Columbia Forestry]. 1997. Just the facts: A review of silviculture and other forestry statistics. <www.for.gov.bc.ca/hfp/forsite/jtfacts/silv3b.htm> (11 Feb. 2000).

BC Fisheries. 1998. British Columbia aquaculture production 1994-96. <www.fisheries.gov.bc.ca/Maff/aquastat.htm> (date).

BCSTATS. 1998a. British Columbia Regional District July 1 population estimates: 1996-97. <www.bcstats.gov.bc.ca/data/pop/pop/cdpop5.htm> (5 Aug. 1998).

BCSTATS. 1998b. British Columbia Regional District population projection 2008-2012. P.E.O.P.L.E. Projection Run 23. <www.bcstats.gov.bc.ca/data/pop/pop/rdproj3.htm> (5 Aug. 1998).

Beauchamp, E.G. and R.P. Voroney. 1994. Crop carbon contribution to the soil with different cropping and livestock systems. J. Soil Water Conser. 49: 205-209.

Behroozi, K, R. Guttman, N. Gruener and H.I. Shuval. 1971. Changes in the motor activity of mice given sodium nitrate drinking solution. *In* Proceedings of a Symposium on Environmental Physiology, Beersheba, Israel. December 1971.

Beklioglu, M. and B. Moss. 1996. Existence of a macrophyte-dominated clear water state over a very wide range of nutrient concentrations in a small lake. Hydrobiol. 337: 93-106.

Bell, S.A., T.W. Ashenden and C.R. Rafael. 1992. Effects of rural roadside levels of nitrogen dioxide on *Polytrichum formosum*. Environ. Pollut. 76: 11-14.

Benedict, H.M and W.H. Breen. 1955. The use of weeds as a means of evaluating the damage caused by air pollution. *In* Proceedings of the 3rd National Air Pollution Symposium, Los Angeles, CA. p. 177-190.

Bennet, J.H. 1975. Acute effects of combinations of SO_2 and NO_2 on plants. Environ. Pollut. 9: 127-132.

Berden, M and S.I. Nilsson. 1996. Influence of added ammonium sulphate and the leaching of aluminium, nitrate and sulphate- a lab experiment. Water Air Soil Pollut. 87: 1-22.

Berge, D. and T. Källqvist. 1998. Biological availability of various P-sources studied in different test systems. Verh. Internat. Verein. Limnol. 26: 2401-2404.

Berger, L. 1989. Disappearance of amphibian larvae in the agricultural landscape. Ecol. Inter. Bull. 17: 65-73.

Beresford, S.A.A. 1985. Is nitrate in the drinking water associated with the risk of cancer in the urban UK? Inter. J. Epidemiol. 14: 57-72.

Besford, R. T. and D.W Hand. 1989. The effects of CO_2 enrichment and nitrogen oxides on some Calvin cycle enzymes and nitrite reductase in glasshouse lettuce. J. Exper. Bot. 40(212): 329-343.

Betcher, R.N. 1997. Rural groundwater quality surveys southern and central Manitoba. *In* Proceedings of Rural Water Quality Symposium, March 25-26, 1997, Winnipeg, MB.

Bicki, T.J., R.B. Brown, M.E. Collins, R.S. Mansell and D.F. Rothwell. 1984. Impact of on-site sewage disposal systems on surface and ground water quality. Report to Florida Department of Health and Rehabilitative Services. Institute of Food and Agricultural Sciences, University of Florida, Gainesville, FL.

Biederbeck, V.O., W.A. Rice, C. van Kessel, L.D. Bailey and E.C Huffman. 1996. Present and future nitrogen gains from legumes in major soil zones of the prairies. *In* Soils & Crops'96. Proceedings of the Soils and Crops Workshop. February 22-23, 1996, University of Saskatchewan, Saskatoon, SK. p. 441-445

Biggs, B.J.F. and M.E. Close. 1989. Periphyton biomass dynamics in gravel bed rivers: the relative effects of flow and nutrients. Freshwat. Biol. 22: 209-231.

Biggs, B.J.F. and R.L. Lowe. 1994. Responses of two trophic levels to patch enrichment along a New Zealand stream continuum. N.Z. J. Mar. Freshwat. Res. 28: 119-134.

Billings, W.H. 1981. Water-associated human illness in northeast Pennsylvania and its suspect association with blue-green algae blooms. *In* The Water Environment: Algal Toxins and Health. *Edited by* W.W. Carmichael. Plenum Press. N.Y. p. 243-256.

Binkley, D. and T.C. Brown. 1993. Forest practices as nonpoint sources of pollution in North America. Water Res. Bull. 29: 729-740.

Bird, D. and V. Mesnage. 1996. Evaluation du bilan annuel de phosphore dans le Lac Heney. Rep. Prep. for the Ministère de l'Environnement et de la Faune and the Association for the Protection of Heney Lake. Groupe de Recherche en Ecologie Aquatique. Université du Québec à Montréal.

Birtwell, I.K., G.L. Greer, M.D. Nassichuk and I.H. Rogers. 1983. Studies on the impact of municipal sewage discharged into an intertidal area within the Fraser River estuary, British Columbia. Can. Tech. Rep. Fish. Aquat. Sci. No. 1170. 55 p.

Black, E.A. and J.R. Forbes. 1997. Overview of effects of elevated nutrients from fish farm wastes on phytoplankton productivity. *In* Report to the provincial assessment review of salmon aquaculture in British Columbia. Department of Fisheries and Oceans. p. 48-57.

Black, S.A., D.N. Graveland, W. Nickolaichuck, D.W. Smith, R.S. Tobin, M.D. Webber and T.R. Brindle. 1984. Manual for Land Application of Treated Municipal Wastewater and Sludge. Environment Canada Report EPS 6-EP-84-1, Ottawa, ON.

Blaustein, A.R., D.B. Wake and W.P. Sousa. 1994. Amphibian declines: Judging stability, persistence, and susceptibility of populations to local and global extinctions. Conser. Biol. 8: 60-71.

Bobbink, R., L. Bik and J.H. Willems. 1988. Effects of nitrogen fertilization on vegetation structure and dominance of *Brachypodium pinnatum* (L.): Beauv. in chalk grassland. Acta. Bot. Neerl. 37:231-242.

Boghen, A.D. 1989. Cold-water aquaculture in Atlantic Canada. Canadian Institute for Research on Regional Development, Moncton, NB.

Bohn, H. L., B. L. McNeal and G. A. O'Connor. 1985. Soil Chemistry, 2nd ed. John Wiley & Sons, New York. 341 p.

Boland J.J., B.P. Anderson, N.H. Brooks, W.M. Eichbaum, L.R. Goldman, D.R.G. Harleman, R.W. Howarth, R.J. Hugget, T.M. Keinath, A.J. Mearns, C.R. O'Melia, L.A. Roesner, J.B. Rose and J.R. Schubel. 1993. Managing Wastewater in Coastal Urban Areas. National Academy Press, Washington, DC. 477 p.

Bolinder, M.A., D.A. Angers and J.P. Dubuc. 1997. Estimating shoot to root ratios and annual carbon inputs in soils for cereal crops. Agric. Ecosys. Environ. 63: 61-66.

Bolinder, M.A., R.R. Simard, S. Beauchemin and K.B. MacDonald. 2000. Risk of water contamination by phosphorus. *In* Environmental sustainability of Canadian agriculture: Report of the agri-environmental indicator project. *Edited by* T. MacRae, C.A.S. Smith and L.J. Gregorich. Research Branch, Policy Branch, Prairie Farm Rehabilitation Administration, Agriculture and Agri-Food Canada.

Bothwell, M.L. 1985. Phosphorus limitation of lotic periphyton growth rates: an intersite comparison using continuous-flow troughs (Thompson River system, British Columbia). Limnol. and Oceanogr. 30: 527-542.

Bothwell, M.L. 1992. Eutrophication of rivers by nutrients in treated kraft pulp mill effluent. Water Pollut. Res. J. Can. 27: 447-472.

Bothwell, M.L., S. Jasper and R.J. Daley. 1989. Phosphorus Control of Algal Production and Biomass in the Thompson River, British Columbia. IWD Scientific Series No. 165. 9 p.

Bothwell, M.L., G. Derksen, R.N. Nordin and J.M. Culp. 1992. Nutrient and grazer control of algal biomass in the Thompson River, British Columbia: a case history of water quality management. *In* Aquatic Ecosystems in Semi-Arid Regions: Implications for Resource Management. *Edited by* R.D. Robarts and M.L. Bothwell. NHRI Symposium Series 7, Environment Canada, Saskatoon, SK. p. 253-266.

Bourke, A.T.C., R.B. Hawes, A. Nielson and N.D. Stallman. 1983. An outbreak of hepatoenteritis (the Palm Island mystery disease) possibly caused by algae intoxication. Toxicon. Suppl. 3: 45-48.

Boutin, C. and B. Jobin. 1998. Intensity of agricultural practices and effects on adjacent habitats. Ecol. Appl. 8: 544-557.

Bradfield, G.E. and G.L. Porter. 1982. Vegetation structure and diversity components of a Fraser estuary tidal marsh. Can. J. Bot. 60: 440-451.

Bradley, W.B., H.F. Eppson and O.A. Beath. 1940. Livestock poisoning by oat hay and other plants containing nitrate. Wyoming Agr. Exp. Bull. 241: 3-20.

Bradstock R., J. Sanders and A. Tegart. 1987. Short-term effects on the foliage of an eucalyptus forest after an aerial application of a chemical fire retardant. Austr. For. 50: 71-80.

Brenton, P. and D. Mellish. 1996. The development of an on-farm manure management program. Final report to Canada/Nova Scotia Agreement on the Agricultural Component of the Green Plan. 29 p. <agric.gov.ns.ca/rs/greenplan/waste/131.html> (9 Oct. 1998).

Bretthauer, R. 1978. Some ecological limits of tolerance to *Ochromonas sociabilis*. Verh. Internat. Verein. Limnol. 20: 1850-1854.

Briggins, D.R. and D.E. Moerman. 1995. Pesticides, nitrate-N and bacteria in farm wells of Kings County, Nova Scotia. Water Qual. Res. J. Can. 30: 429-442.

Brisbin, P.E. 1995. Agricultural Nutrient Management in the Lower Fraser Valley. Prepared for BC Ministry of Environment, Land and Parks, BC Ministry of Agriculture, Fisheries and Food, Environment Canada and Fisheries and Oceans Canada. DOE FRAP Report 1995-27.

Broderius, S., R. Drummond, J. Fiandt and C. L. Russom. 1985. Toxicity of ammonia to early life stages of the smallmouth bass at four pH values. Environ. Toxicol. Chem. 4: 87-96.

Broitman, S.A., H. Vélez and J.J. Vital. 1981. A possible role of iron deficiency in gastric cancer in Columbia. Adv. Exp. Med. Biol. 135: 155-167.

Brooks, W.P. and G.A. Codd. 1987. Distribution of *Microcystis aeruginosa* peptide toxin and interactions with hepatic microsomes in mice. Pharm. Toxicol. 60: 187-191.

Brownell, C.L. 1980. Water quality requirements for first feeding in marine fish larvae. I. Ammonia, nitrite, nitrate. J. Exper. Mar. Biol. Ecol. 44: 269-284.

Brownlee, B.G., D.S. Painter and R.J. Boone. 1984. Identification of taste and odour compounds from western Lake Ontario. Water Pollut. Res. J. Can. 19: 111-118.

Bruggink, W.T., H.G. Wolting, J.H.A. Dassen and V.M. Bus. 1988. The effect of nitric oxide fumigation at two CO_2 concentrations on net photosynthesis and stomatal resistance of tomato (*Lycopersicon lycopersicum* L. cv. Abunda). New Phytol. 110: 185-191.

Brunskill, G.J. 1973. Rates of supply of nitrogen and phosphorous to Lake Winnipeg, Manitoba, Canada. Int. Ver. Theor. Angew. Limnol. Verh. 18: 1755-1759.

Brunskill, G.J., S.E.M. Elliot and P. Campbell. 1980. Morphometry, hydrology, and watershed data pertinent to the limnology ofLake Winnipeg. Can. Fish. Serv. MS Rep. No. 1556.

Brylinsky, M., G.R. Daborn, D.J. Wildish, M.J. Dadswell, P.W. Hicklin, C.D. Duncan, W.T. Stobo, M.W. Brown and S.D. Kraus. 1996. The biological environment of the Bay of Fundy. *In* Bay of Fundy Issues: A Scientific Overview. Workshop proceedings. *Edited by* J.A. Percy, P.G. Wells and A.J. Evans. Wolfville, N.S. Jan 29 to Feb 1, 1996. Environment Canada- Atlantic Region Occasional Report No. 8. Environment Canada, Sackville, NB. p. 63-101.

Buckley, J.E. 1978. Acute toxicity of un-ionized ammonia to fingerling coho salmon. Prog. Fish. Cult. 40: 30-32.

Burkholder, J.M., E.J. Noga, C.W. Hobbs, H.B. Glasgow, Jr. and S.A. Smith. 1992. New "phantom" dinoflagellate is the causative agent of major estuarine fish kills. Nature 358:407-410; Nature 360:768.

Burns, N.M. 1976. Nutrient budgets for Lake Erie, 1970. J. Fish Res. Bd. Can. 33:520-536.

Busch, R.H., R.L. Buschbom, W.C. Cannon, K.E. Lauhala, F.J. Miller, J.A. Graham and L.G. Smith. 1986. Effects of ammonium nitrate aerosol exposure on lung structure of normal and elastase-impaired rats and guinea pigs. Environ. Res. 39: 237-252.

Buth, J.L., H. Rohde and R. Butler. 1992. Groundwater quality assessment of the Assiniboine Delta Aquifer. Manitoba Agriculture, Carman, MB, 12 p.

CAESA [Canada-Alberta Environmentally Sustainable Agriculture Agreement]. 1998. Agricultural impacts on water quality in Alberta; An initial assessment. Lethbridge, AB.

CAIA [Canadian Aquaculture Industry Alliance]. 1998. Industry profile information. <www.aquaculture.ca> (27 April 1998).

Camargo, J.A. and J.V. Ward. 1992. Short-term toxicity of sodium nitrate ($NaNO_3$) to non-target freshwater invertebrates. Chemosp. 24: 23-38.

Cameron, R. and B. Obee 1981. Special Report- The Lower Fraser: How well are we looking after B.C.'s front door? British Columbia Ministry of Environment. Wildlife Rev. 10: 3-12.

Campbell, C.A., R. De Jong and R.P. Zentner. 1984. Effect of cropping, summerfallow and fertilizer nitrogen on nitrate-nitrogen lost by leaching on a brown chernozemic loam. Can. J. Soil. Sci. 64: 61-74.

Campbell, C.A., R.P. Zentner, F. Selles and O.O. Akinremi. 1993. Nitrate leaching as influenced by fertilization in the brown soil zone. Can. J. Soil Sci. 73: 387-397.

Campbell, H.W. and M.D. Webber. 1997. Biosolids management in Canada: Current practice and future trends. Presented to the Specialty Conference on Management and Fate of Toxic Organics in Sludge Applied to Land, Apr. 30-May 2, 1997, Copenhagen, Denmark.

Campbell, J. B., A. N. Davis and P.J. Myhr. 1954. Methemoglobinemia of livestock caused by high nitrate contents of well water. Can J. Comp. Med. 18: 93-101.

Canfield, D.E. Jr. 1983. Prediction of chlorophyll a concentrations in Florida lakes: the importance of phosphorus and nitrogen. Water Resour. Bull. 19: 255-262.

Canfield, D.E. Jr. and M.V. Hoyer. 1988. Influence of nutrient enrichment and light availability on the abundance of aquatic macrophytes in Florida streams. Can. J. Fish. Aquat. Sci. 45: 1467-1472.

Cape, J.N., I.D. Leith, D. Fowler, M.B. Murray, L.J. Sheppard, D. Eamus and R.H.F. Wilson. 1991. Sulphate and ammonium in mist impair the frost hardening of red spruce seedlings. New Phytol. 118: 119-126.

Capron, S.J.M. 1989. The effect of oxides of nitrogen and CO_2 enrichment on photosynthesis and growth of lettuce (*Lactuca sativa* L.). New Phytol. 111: 473-481.

Capron, S.J.M, T.A. Mansfield and D.W. Hand. 1991. Low temperature-enhanced inhibition of photosynthesis by oxides of nitrogen in lettuce (*Lacusa sativa* L). New Phytol. 118: 309-313.

Caraco, N.F. 1993. Disturbance of the phosphorus cycle: a case of indirect effects of human activity. Trends Ecol. Evolut. 7: 51-54.

Caraco, N.F. 1995. Influence of human populations on P transfers to aquatic systems: a regional scale study using large rivers. *In* Phosphorus in the Global Environment. *Edited by* H. Tiessen. John Wiley, New York, NY. p. 235-244.

Cargill, S.M. and R.L. Jeffries. 1984. Nutrients limitation on primary productivity in a sub-arctic salt marsh. J. Appl. Ecol. 21: 257-268.

Carignan, R., P. D'Arcy and S. Lamontagne. 2000. Comparative impacts of fire and forest harvesting on water quality in Boreal Shield Lakes. Can. J. Fish. Aquat. Sci. 57 (Suppl. 2): 105-117.

Carmichael, V., M. Wei and L. Ringham. 1995. Fraser Valley groundwater monitoring program final report.

Carmichael, W.W. 1986. Algal toxins. Adv. Bot. Res. 12: 47-101.

Carmichael, W.W. 1988. Toxins in freshwater algae. *In* Handbook of Natural Toxins. VOL. III. Marine Toxins and Venoms. *Edited by* A.T. Tu. Dekker, N.Y. p. 121-127.

Carmichael, W.W. 1989. Freshwater cyanobacteria (blue-green algae) toxins. *In* Natural Toxins. Proceedings from the Ninth World Congress on Animal, Plant and Microbial Toxins. *Edited by* C.L. Ownby and G.V Odell. Pergamon Press. N.Y. p. 3-16.

Carmichael, W. 1992. Cyanobacteria secondary metabolites - the cyanotoxins. J. Appl. Bacteriol. 72:445-459.

Carmichael, W.W. 1994. The toxins of cyanobacteria. Sci. Amer. (Jan. 1994): 78-86.

Carmichael, W.W., D.F. Biggs and P.R. Gorham. 1975. Toxicology and pharmacological action of *Anabaena flos-aquae* toxins. Sci. 187: 542-544.

Carmichael, W.W. and P.R. Gorham. 1978. Anatoxins from clones of *Anabaena flos-aquae* isolated from lakes of Western Canada. Mitt. Internat. Verein. Limnol. 21: 285-295.

Carmichael, W.W., C.L.A. Jones, N.A. Mahmood and W.C. Theiss. 1985. Algal toxins and water-based diseases. CRC Crit. Rev. Environ. Contam. 15: 275-313.

Carpenter, S.R., N.F. Caraco, D.L. Correll, R.W. Howarth, A.N. Sharpley and V.H. Smith. 1998. Nonpoint pollution of surface waters with phosphorus and nitrogen. Ecol. Appl. 8: 559-568.

Carr, G.M. and P.A. Chambers. 1998. Spatial and temporal patterns in nutrients and algal abundance in Alberta rivers. Report prepared for the Prairie Provinces Water Board, Regina, SK.

Carvalho, L. and B. Moss. 1995. The current status of a sample of English sites of special scientific interest subject to eutrophication. Aquat. Conser. Mar. Freshwat. Eco. 5: 191-204.

Catalla, W.J. 1993. Ecotoxicology and wetland ecosystems: Current understanding and future needs. Environ. Toxicol. Chem. 12: 2209-2224.

Cattaneo, A., T. Kerimian, M. Roberge and J. Marty. 1997. Periphyton distribution and abundance on substrata of different size along a gradient of stream trophy. Hydrobiol. 354: 101-110.

Cavanagh, D.G. and J.B. Morris. 1987. Mucus protection and airway peroxidation following nitrogen dioxide exposure in the rat. J. Toxicol. Environ. Hlth. 22: 313-328.

CBCL Limited. 1996. Atlantic Canada standards and guidelines manual for the collection, treatment, and disposal of sanitary sewage. Prepared for Environment Canada, Dartmouth, NS.

CCC [Composting Council of Canada]. 1998. Setting the standard: A summary of compost standards in Canada. <www.compost.org/standard.html> (19 Oct. 1998).

CCME [Canadian Council of Ministers of the Environment]. 1999. Environmental quality guidelines. CCME Winnipeg, MB.

CCME [Canadian Council of Ministers of the Environment]. 2000. Canadian water quality guidelines for the protection of aquatic life for ammonia. Winnipeg, MB.

CCREM [Canadian Council of Resource and Environment Ministers]. 1987. Canadian Water Quality Guidelines. Prepared by the Task Group on Water Quality Guidelines of the Canadian Council of Resource and Environment Ministers. Ottawa, ON.

CEEP [Centre Européen d'Études des Polyphosphates]. 1998. Phosphates: A Sustainable Future in Recycling. Produced by CEEP, a sector group of CEFIC (European Chemical Industry Council), Brussels, Belgium.

Chalmers, A., C Kershaw, P. Leech. 1990. Fertilizer use on farm crops in Great Britain. Outlook Agric. 19(4): 269-278.

Chalmers, A.G. 1979. The effects of fertilization on nitrogen distribution in a *Spartina alterniflora* salt marsh. Est. Coast. Shelf Sci. 8: 327-337.

Chambers, P.A. 1989. Reconnaissance study of the Qu'Appelle Lakes. Report on the September 1989 evaluation of aquatic macrophyte growth in the Qu'Appelle Lakes. N.H.R.I Contribution No. 89090.

Chambers, P.A. 1996. Nutrient enrichment in the Peace, Athabasca and Slave Rivers: Assessment of present conditions and future trends. Northern River Basins Study, Edmonton, AB. Synthesis Report No. 4. 107 p.

Chambers, P.A., A-M. Anderson, C. Bernard, L..J. Gregorich, B. McConkey, P.H. Milburn, J. Painchaud, N.K. Patni, R.R. Simard and L.J.P. van Vliet. 2000. Surface water quality. *In*: The health of our waters: Toward sustainable agriculture in Canada. *Edited by* D.R. Coot and L..J. Gregorich. Agriculture and Agri-Food Canada publication 2020/E. Ottawa, ON. 173 p.

Chambers P.A., A.R. Dale, G.J. Scrimgeour and M.L. Bothwell. 2000. Nutrient enrichment of northern rivers in response to pulp mill and municipal discharges. J. Aquatic Ecosystem Stress Recovery 8: 53-66.

Champ, M.A., J.T. Lock, C.D. Bjork, W.G. Klussmann and J.D. McCullough Jr. 1973. Effects of anhydrous ammonia on a central Texas pond and a review of previous research with ammonia in fisheries management. Trans. Amer. Fish. Soc. 102: 73-82.

Chang, C. and T. Entz. 1996. Nitrate leaching losses under repeated cattle feedlot manure applications in southern Alberta. J. Environ. Qual. 25: 145-153.

Chapman, V.J. 1960. Salt marshes and salt deserts of the world. Hill/Leonard Books, London. 378 p.

Chapman, V.J. 1974. Salt Marshes and salt deserts of the world, 2nd Suppl. reprint ed. Verlag Von J. Cramer. 392 p.

Charlton, M.N. 1980. Hypolimnion oxygen consumption in lakes: discussion of productivity and morphometry effects. Can. J. Fish. Aquat. Sci. 37: 1531-1539.

Charlton, M N. 1994. The case for research on the effects of zebra mussels in Lake Erie: visualization of information from August and September 1993. J. Biol. Sys. 2: 467-480.

Charlton, M. N., R. Le Sage and J.E. Milne. 1998. Lake Erie in transition: the 1990s. *In*. State of Lake Erie (SOLE) - Past , Present and Future. *Edited by* M. Munawar, T. Edsall and I.F. Munawar. Ecovision World Mongraph Series, 1999 Backhuys Publishers, Leiden, The Netherlands, p. 97-123.

Chartrand, J., P. Levallois, D. Gauvin, S. Gingras, J. Rouffignat and M-F. Gagnon. 1999. La contamination de l'eau potable par les nitrates a l'Île d'Orléans. Vecteur Environnement, 32: 37-46.

Chemonics. 1991. Summary of fire chemical toxicity test: Fire-Trol LCA-R (raffinate). Data submitted to U.S. Forest Service. Phoenix, AZ.

Chen, J., W.J. Chen, J. Liu, J. Cihlar and S. Gray. 2000a. Annual carbon balance of Canada's forest during 1895-1996. Global Biogeochem. Cy. 14: 839-849..

Chen, W.J., J. Chen and J. Cihlar. 2000b. An integrated terrestrial ecosystem carbon-budget model based on changes in disturbance, climate, and atomspheric chemistry. Ecol. Model. 135: 55-79.

Cherry, J.A. and R.A. Rapaport. 1994. Chemical fate and transport in a domestic septic system: A case study.- Editorial. Environ. Toxicol. Chem. 13: 181-182.

Chételat, J., F.R. Pick, A. Morin and P.B. Hamilton. 1999. Periphyton biomass and community composition in rivers of different nutrient status. Can. J. Fish. Aquat. Sci. 56: 560-569.

Chiaudani, G. and Premazzi, G. 1988. Appraisal of the possible methods of combating the threat of eutrophication in community waters. Commission of the European Communities (CEC, Brussels).

Cho, C.Y. and D.P. Bureau. 1997. Reduction of waste output from salmonid aquaculture through feeds and feeding. The Prog. Fish Cult.: 59: 155-160.

Cho, C.Y., J.D. Hynes, K.R. Wood and H.K. Yoshida. 1994. Development of high-nutrient-dense, low-pollution diets and prediction of aquaculture wastes using biological approaches. Aquacult. 124: 293-305.

Chorus, I. and J. Bartram. 1999. Toxic Cyanobacteria in Water: A guide to their public health consequences, monitoring and management. E and FN Spon, London UK. 432 p.

Chow C., J. House, R. Velzeboer, M Drikas, M. Burch and D. Steffensen. 1998. The effect of ferric chloride flocculation on cyanobacterial cells. Wat. Res. 32:808-814.

Chow-Fraser, P. 1998. A conceptual ecological model to aid restoration of Cootes Paradise Marsh, a degraded coastal wetland of L. Ontario, Canada. Wetland Ecol. Mgmt. 6: 43-57.

Chow-Fraser, P. and L. Lukasik. 1995. Community participation in the restoration of a Great Lakes coastal wetland. Restor. Mgmt. Notes. 13: 183-189.

Chow-Fraser, P., B. Crosbie, D. Bryant and B. McCarry. 1996. Potential contribution of nutrients and PAHs from the creeks of Cootes Paradise Marsh. Wat. Qual. Res. J. Can. 31: 485-503.

Chow-Fraser, P., V. Lougheed, V. LeThiec, B. Crosbie, L. Simser and J. Lord. 1998. Long-term response of the biotic community to fluctuating water levels and changes in water quality in Cootes Paradise Marsh, a degraded coastal wetland of L. Ontario. Wetland Ecol. Mgmt. 6: 19-42.

Clague, J.J. and J.L. Luternauer. 1982. Late Quaternary sedimentary environments, southwestern B.C. Guidebook 30A, 11th International congress on sedimentology. McMaster University, Hamilton, ON 167 p.

Clark, R.B. 1982. Nutrient solution growth of sorghum and corn in mineral nutrient studies. J. Plant Nutr. 5: 1039-1057.

Clemens, W.A. 1935. Red "water bloom" in British Columbian waters. Nature 135: 473.

Clough, P.W.L. 1983. Nitrates and gastric carcinogenesis. Minerals Environ. 5: 91-104.

Codd, G.A., C. Edwards, K.A. Beattie, W.M. Barr and G.J. Gunn. 1992. Fatal attraction to cyanobacteria? Nature 359: 110-111.

Colt, J. and G. Tchobanogious. 1976. Evaluation of the short-term toxicity of nitrogenous compounds to channel catfish, *Ictalurus punctatus*. Aquacult. 8: 209-224.

Comly, H.H. 1945. Cyanosis in infants caused by nitrates in well water. J. Amer. Med. Assoc.129: 112-116.

Cook, W.O., V.R. Beasley, R.A. Lovell, A.M. Dahlem and S.B. Hooser. 1989. Consistent inhibition of peripheral cholinesterases by neurotoxins from the freshwater cyanobacterium *Anabaena flos-aquae*: Studies of duck, swine, mice and a steer. Environ. Toxicol. Chem. 8: 915-922.

Cooke, S.E. and E.E. Prepas. 1998. Stream phosphorus and nitrogen export from agricultural and forested watersheds on the Boreal Plain. Can. J. Fish. Aquat. Sci. 55: 2292-2299.

Coon, R.A., R.A. Jones, L.J. Jenkins and J. Siegil. 1970. Animal inhalation studies on ammonia, ethylene glycol, formaldehyde, dimethylamine, and ethanol. Toxicol. Appl. Pharm. 16: 646-655.

Corre, W.J. and T. Breimer. 1979. Nitrate and nitrite in vegetables. Centre for Agricultural Publishing and Documentation, Wageningen, The Netherlands. 88 p.

Cote, B., W.H. Hendershot and I. O'Halloran. 1993. Response of sugar maple to seven types of fertilization in southern Quebec: growth and nutrient status. *In* Forest decline in the Atlantic and Pacific Region. *Edited by* R.F. Huettl and D. Mueller-Dombois. Springer-Verlag Berlin, p 162-174.

Cote, B. and R. Ouimet. 1996. Decline of the maple-dominated forest in southern Quebec: Impact of natural stresses and forest management. Environ. Rev. 4: 133-148.

Cowley, G.D. and D.F. Collings. 1977. Nitrate poisoning. Vet. Rec. 101: 305-306.

Cox, R.M., G. Lemieux and M. Lodin. 1996. The assessment and condition of Fundy white birches in relation to ambient exposure to acid marine fogs. Can. J. For. Res. 26: 682-688.

Craun, G.F., D.G. Greathouse and D.H. Gunderson. 1981. Methaemoglobin levels in young children consuming high nitrate well water in the United States. Inter. J. Epidemiol. 10: 309-321.

Cronan C.S. and D.F. Grigal. 1995. Use of calcium/aluminum ratios as indicators of stress in forest ecosystems. J. Environ. Qual. 24: 209-226.

Cross, S.F. 1990. Benthic impacts of salmon farming in British Columbia. Vol. 1, Summary report. Prepared by Aquametrix Ltd., Sidney, B.C. for Ministry of Environment, Lands and Parks, Province of B.C. *Cited in* DFO [Department of Fisheries and Oceans]. 1997. Report to the provincial assessment review of salmon aquaculture in British Columbia.

Crowder, A. 1991. Acidification, metals and macrophytes. Environ. Pollut. 71: 171-203.

Crowe, A., C.J. Ptacek, R. McCrea, P.A. Huddart, F.J. Longstaffe, J.P. Coakley, D.L. Thompson, T. Mayer and S.G. Shikaze. 1998. An overview of Environment Canada's groundwater research and monitoring activities at Point Pelee National Park, Ontario. *In* Proceedings of the Parks Research Forum of Ontario, 2nd Annual General Meeting, February 5-6, 1998, Peterborough, ON. p. 225-238.

Culp, J.M., H.R. Hamilton, A.J. Sosiak and R.W. Davies. 1992. Longitudinal zonation of the biota and water quality of the Bow River system in Alberta, Canada. *In* Water Quality in North American River Systems. *Edited by* C.D. Becker and D.A. Neitzel. Battelle Press, Ohio, U.S.A. 304 p.

D'Arcy, P. and R. Carignan. 1997. Influence of catchment topography on water chemistry in southeastern Quebec Shield lakes. Can. J. Fish. Aquat. Sci. 54: 2215-2227.

Dederen, L.H.T. 1992. Marine eutrophication in Europe: Similarities and regional differences in appearance. Sci. Tot. Environ. 6 (Suppl.): 663-672.

Delisle, F., S. Gariépy and Y. Bédard. 1998. Basin versant de la rivière Yamaska : l'activité agricole et ses effets sur la qualité de l'eau. Ministère de l'Environnement et de la Faune du Québec et Saint-Laurent Vision 2000. 124 p.

Desjardins, R.L. and J.C. Keng. 1999. Nitrous oxide emissions from agricultural sources in Canada. *In* Proceedings of the International Workshop on Reducing Nitrous Oxide Emissions from Agroecosystems, Banff, AB, March 3-5, 1999. *Edited by* R.L. Desjardins, J.C. Keng and K. Haugen-Kozyra. Agriculture and Agri-Food Canada, Research Branch; Alberta Agricuture Food and Rural Development, Conservation and Development Branch.

DFO/DEMR/DOE Science Advisory Committee on Halifax Harbour Sewage Disposal. 1989. Halifax Harbour marine environmental quality. *In* Investigations of Marine Environmental Quality in Halifax Harbour. *Edited by* H.B. Nicholls. Can. Tech. Rep. Fish. Aquat. Sci. No. 1693. v. + 83 p.

DFO [Department of Fisheries and Oceans]. 1997. Report to the provincial assessment review of salmon aquaculture in British Columbia.

DFO [Department of Fisheries and Oceans]. 1998. Preliminary 1996 Canadian aquaculture production statistics. <www.dfo-mpo.gc.ca/communic/statistics/aquacult/AQUA96.htm> (26 Mar. 1998).

DFO [Department of Fisheries and Oceans]. 1999. More about aquaculture. <www.dfo-mpo.gc.ca/ocad-bcda/more.html> (29 Nov. 1999).

Dillenberg, H.O. and M.K. Dehnel. 1960. Toxic water-bloom in Saskatchewan, 1959. Can. Med. Ass. J. 83: 1151-1154.

Dillon, P.J. and F.H. Rigler. 1974. The phosphorus-chlorophyll relationship in lakes. Limnol. Oceanogr. 19: 767-773.

Dillon, P.J. and F.H. Rigler . 1975. A simple method for predicting the capacity of a lake for development based on lake trophic status. J. Fish. Res. Bd. Can. 32: 1519-1531.

Dillon, P.J., L.A. Molot and W.A. Scheider. 1991. Phosphorus and nitrogen export from forested stream catchments in central Ontario. J. Environ. Qual. 20: 857-864.

Dillon, P.J., K.H. Nicholls, W.A. Scheider, N.D. Yan and D.S. Jeffries. 1986. Lakeshore Capacity Study. Trophic status. Ontario Ministry of Municipal Affairs, Queen's Printer for Ontario.

Dixit, A.S., R.I. Hall, P.R. Leavitt, R. Quinlan and J.P. Smol. 2000. Effects of sequential depositional basins on lake response to urban and agricultural pollution: A palaeoecological analysis of the Qu'Appelle Valley, Saskatchewan, Canada. Freshwat. Biol. 43: 319-338.

Dixon, R.W.J. 1994. Stormwater Quality and Loading to the Glenmore Reservoir. The City of Calgary, Engineering and Environmental Services Department.

DND [Department of National Defence, Transport Canada, Environment Canada]. 1998. An evaluation and interpretation of stormwater quality data from federal facilities. Ottawa, ON. 50 p.

Dodd, K.T. and D.R. Gross 1980. Ammonia inhalation toxicity in cats. Arch. Environ. Hlth. 35:14-16.#4

Dodds, W.K. V.H. Smith and B. Zander. 1997. Developing nutrient targets to control benthic chlorophyll levels in streams: a case study of the Clark Fork River. Water. Res. 31: 1738-1750.

DOE [Department of Environment]. 1989. Phosphorus concentration regulations. Extract Canada Gazette, Part III, Nov. 8, 1989. No. 23, vol. 123. SOR/DORS/89-501.

Dorcey, A.H.J. and J.R. Griggs. 1991. Water in sustainable development: Exploring our common future in the Fraser River Basin. Vol. II. Westwater Research Centre. Vancouver, BC, Canada.

Dorcey, A.H.J., T.G. Northcote and D.V. Ward. 1978. Are the Fraser marshes essential to salmon? Westwater Lectures, University of British Columbia, No. 1. July 1978. 29 p.

Druckrey, H., D. Steinhoff, H. Beuthner,. H. Schneider and P. Klärner. 1963. Screening of nitrite for chronic toxicity in rates. Arzneimittel-Forsch. 13: 320-323.

Duarte, C.M. 1995. Submerged aquatic vegetation in relation to different nutrient regimes. Ophelia 41: 87-112.

Duarte, C.M. and D.E. Canfield Jr. 1990. Macrophyte standing crop and primary productivity in some Florida spring-runs. Water Resour. Bull. 26: 927-934.

Duchesne, L.C. 1994. Ecological impact of fire protection. *In* Foam Applications for Wildland and Urban Fire Management. *Edited by* National Wildfire Coordinating Group and Canadian Committee on Forest Fire Management. International wildland fire foam symposium and workshop, Thunder Bay, ON. May 3-7, 1994. 15 p.

Dueck, T.A. and J. Elderson. 1992. Influences of NH_3 and SO_2 on the growth and competitive ability of *Arnica montana* L. and *Viola canina* L. New Phytol. 122: 507-514.

Dueck, T.A., F.G. Dorel, R. Ter Horst and L.J.M. Van der Eerden. 1990. Effects of ammonia, ammonium, sulphate, and sulphur dioxide on the frost sensitivity of Scots pine (*Pinus sylvestris* L.). Water Air Soil Pollut. 54: 35-49.

EC [European Community]. 1999a. Community legislation in force <europa.eu.int/eur-lex/en/lif/dat/1991/en_391L0271.html> (12 Nov. 1999).

EC [European Community]. 1999b. Developments of the water framework directive. <europa.eu.int/water/water-framework/index_en.html> (19 Feb. 2000).

Ecobichon, D.J., R. Hicks, M.C. Allen and R. Albert. 1996. Ground water contamination in rural New Brunswick. *In* Agriculture and Groundwater Quality Workshop. Agricultural Advisory Committee on the Environment, April 1996, Grand Falls, NB, 90 p.

Edmonson, W.T. 1969. Eutrophication in North America. *In* Eutrophication: Causes, Consequences, Correctives. *Edited by* National Academy of Science Washington, D.C. p. 124-149.

Ehrenfeld, J.G. 1983. The effects of changes in land-use on swamps of the New Jersey Pines Barrens. Biol. Conser. 25: 353-375.

Ehrlich, P.R. and A. Ehrlich. 1981. Extinction: The causes and consequences of the disappearance of species. Random House Publishing, N.Y. 305 p.

Einen, O., I. Holmefjord, T. Asgard and C. Talbot. 1995. Auditing nutrient discharges from fish farms: Theoretical and practical consideration. Aqua. Res. 26: 701-713.

El-Bahri, L., J. Belguith and A. Blouin. 1997. Toxicology of nitrates and nitrites in livestock. Comp. Con. Ed. Prac. Vet. 19: 643-649.

Eloff, J.N. and A.J. Van Der Westhuizen. 1981. Toxicology studies on *Microcystis*. *In* The Water Environment: Algal Toxins and Health. *Edited by* W.W. Carmichael. Plenum Press. N.Y. p. 343-364.

Elwood, J.W., J.D. Newbold, A.F. Trimble and R.W. Stork. 1981. The limiting role of phosphorous in a woodland stream ecosystem: effects of P enrichment on leaf decomposition and primary producers. Ecology 62: 146-158.

Emerick, R.J. 1974. Consequence of high nitrate levels in feed and water supply. Fed. Proc. 33: 11-83.

Environment Canada. 1983. National inventory of sources and emissions of phosphorus (1978). Catalogue No. EPS 3-EP-83-7, Ottawa, ON.

Environment Canada. 1984. Ammonia: Environmental and technical information for problem spill. Environmental Protection Service. Ottawa, ON. 158 p.

Environment Canada. 1993a. Environmental Choice Program Briefing Note – laundry detergents. Ottawa. ON.

Environment Canada. 1993b. Environmental Choice Program Information Sheet on Automatic Dishwashing Detergents. Ottawa. ON.

Environment Canada. 1993c. Environmental Choice Program Environmental Choice Guideline ECP-33-1 General Purpose Cleaners. Ottawa. ON.

Environment Canada. 1995. Guidelines for algae and aquatic vascular plant biomass in Canadian streams. Draft. January 1995. 26 p.

Environment Canada. 1996a. Municipal water use database. Ottawa, ON.

Environment Canada. 1996b. National pollutant release inventory summary report 1996. Catalogue # En40-495/1-1996E, Ottawa, ON.

Environment Canada. 1996c. Perspectives in solid waste management in Canada: An assessment of the physical, economic and energy dimensions of solid waste management in Canada. Hazardous Waste Branch. March 1996, EPS 2/UP/2.

Environment Canada. 1997a. Canada's second national report on climate change. Catalogue # En21-125/1997E, Ottawa, ON.

Environment Canada. 1997b. Problem formulation for ammonia in the aquatic environment. Canadian Environmental Protection Act. Priority substances list 2. Version 5.0. Edmonton, AB. 50 p.

Environment Canada. 1998a. Aquatic Science: Fraser River Action Plan Final Report. Vancouver, BC, Canada.

Environment Canada. 1998b. Endocrine disrupters and hog manure. Sci. Environ. Bull. Nov./Dec. p. 1.

Environment Canada. 1998c. Water quality in the prairie ecozone. <www.mb.ec.gc.ca/ENGLISH/WATER/WQPRAIRIE/> (December, 1998).

Environment Canada. 1999a. 1995 NOx Emissions by province (tonnes). <www.ec.gc.ca/pdb/cac/cacdoc/1995e/nox95.htm> (February, 1999).

Environment Canada. 1999b. Canadian Environmental Protection Act Priority Substances List II: Supporting Ddcument for ammonia in the aquatic environment. Draft.

Environment Canada. 1999c. Canadian Great Lakes Remedial Action Plans Update. <www.cciw.ca/glimr/raps/cdn-intro.html> (15 Mar. 1999).

Environment Canada. 1999d. White haze over the southern region of the lower Fraser Valley. Sci. Environ. Bull. No. 12, May/June 1999.

Environment Canada. 2000a. Environmental sustainability: Canada's agricultural soils. National Environmental Indicator Series, SOE Bulletin No. 2000-1 Ottawa, ON: Environment Canada, Indicators and Assessment Office, State of the Environment Reporting Program.

Environment Canada. 2000b. Environmental Sustainability: Canada's Agricultural Soils. National Environmental Indicator Series, SOE Technical Supplement No. 2000-1 Ottawa, ON: Environment Canada, Indicators and Assessment Office, State of the Environment Reporting Program.

Environment Canada and BCELP [British Columbia Ministry of Environment Lands and Parks]. 1992. State of the environment for the lower Fraser River Basin, Canada. Victoria, BC. 83 p.

Environment Canada and BCELP [British Columbia Environment, Lands and Parks]. 1993. State of the environment report for British Columbia. Victoria, BC and Ottawa, ON.

Environment Canada and PEI Department of Environment. 1990. Brudenell-Cardigan Rivers Complex: Watershed Activities. Environment Canada Inland Waters Directorate. Atlantic Region. Dartmouth N.S., Canada. 27 p.

Epifanio, L.E. and R.F. Srna. 1975. Toxicity of ammonia, nitrate ion, and orthophosphate to *Mercenario mercenario* and *Crassostrea virgincia*. Mar. Biol. 33: 241-246.

Epp, P.F. 1984. British Columbia Ministry of Environment manual 5, soil constraints for septic tank effluent absorption. Kelowna, BC.

EPS [Environmental Protection Service]. 1985. Urea, environmental and technical information for problem spill. Technical Services Branch. Ottawa, ON.

Evans, T. 1998. Assessing the risks of recycling to land. Water Clean Environ. Sept. 98 7:27-30.

EVS Consultants. 1992. Sediment and Related Investigations off the Macaulay and Clover Point Sewage Outfalls. EVS Consultants, Vancouver, BC.

Falconer, I.R. 1991. Tumour promotion and liver injury caused by oral consumption of cyanobacteria. Environ. Toxic. Water Qual. Inter. J. 6: 177-184.

Falconer, I.R. and T.H. Buckley. 1989. Tumour promotion by *Microcystis* sp. a blue green algae occurring in water supplies. Med. J. Austr. 1: 351-372.

Falconer, I.R., A.M. Beresford and M.T.C. Runner. 1983. Evidence of liver damage by toxin from a bloom of blue-green alga, *Microcystis aeruginosa*. Med. J. Austr. 1: 511-514.

Falconer, I.R., J.V. Smith, A.R.B. Jackson, A. Jones and M.T.C. Runner. 1988. Oral toxicity of a bloom of the cyanobacterium *Microcystis aeruginosa* administered to mice over periods up to one year. J. Toxicol. Environ. Hlth. 24: 291-305.

Falconer, I.R., M.D. Burch, D.A. Steffensen, M. Choice and O.R. Coverdale. 1994. Toxicity of the blue-green alga (Cyanobacterium) *Microcystis aeruginosa* in drinking water to growing pigs, as an animal model for human injury and risk assessment. Environ. Toxicol. Water Qual. Intern. J. 9: 131-139.

FAO [Food and Agriculture Organization of the United Nations]. 1999. FAOSTAT Agriculture data. <apps.fao.org/cgi-bin/nph-db.pl?subset=agriculture> (28 Jan. 1999).

Fangmeijer, A.A., L. Hadwiger-Fangmeijer, L.J. Van der Eerden and H.J. Jäger. 1994. Effect of atmospheric ammonia on vegetation: A review. Environ. Pollut. 86: 43-82.

Fawell, J.F., C.P. James and H.A. James. 1994. Toxins from blue-green algae toxicological assessment of microcystin-LR and a method for its determination in water. Report No. FR 0359/2/DOE.335812, Water Research Centre. Medmenham, UK. 46 p.

Feller M.C. and J.P. Kimmins 1984. Effects of clearcutting and slash burning on streamwater chemistry and watershed nutrient budgets in southwestern British Columbia. Water Resour. Res. 20: 29-40.

Fernald. M.L. 1942. Misinterpretations of Atlantic coastal plain species. Rhodora 44: 238-246.

Fitzgerald, D.A., D.A. Kiely, R.D. Neilson, S. Shaw, R.J. Audette, M. Prior, E. Ashton and E. Allison. 1997. Alberta farmstead water quality survey. Canada-Alberta Environmentally Sustainable Agriculture Agreement, Lethbridge, AB.

Flaten, D.N. and R.A. Hedlin. 1988. Impact of technology on crop productivity in Western Canada. *In* Proceedings of the 34th Annual Meeting of the Canadian Soil Science Society, Calgary, AB.

Fontenot, Q.C., J.J. Isely and J.R Tomasso. 1998. Acute toxicity of ammonia and nitrite to shortnose sturgeon fingerlings. Prog. Fish. Cult. 60: 315-318.

FP-EARP [Federal-Provincial Environmental Assessment Review Panel]. 1993. Halifax Harbour Cleanup Project. Hull, QC. 79 p.

Fraser, A.S. 1987. Tributary and point source total phosphorus loading to Lake Erie. J. Great Lakes Res. 13: 659-666.

Fraser, P. 1985. Nitrates: epidemiological evidence. IARC Sci. Publ. 65: 183-202.

Fraser, P. and C. Chilvers. 1981. Health aspects of nitrate in drinking water. Sci. Total Environ. 18: 103-116.

Freedman, B., V. Zelazny, D. Beaudette, T. Fleming, S. Flemming, G. Forbes, J.S. Gerrow, G. Johnson and S. Woodley. 1996. Biodiversity implications of changes in the quantity of dead organic matter in managed forest. Environ. Res. 4(3): 238-265.

French, T.D. and P.A. Chambers. 1995. Nitrogen and phosphorus in the Upper Fraser River in relation to point and diffuse source loadings. DOE FRAP 1995-09.

Friedland, A.J., R.A. Gregory, L. Kärenlampi and A.J. Johnson. 1984. Winter damage to foliage as a factor in red spruce decline. Can. J. For. Res. 14: 963-965.

Fulrani, A.M.C., R.B. Clark, C.Y. Sullivan and J.W. Maranville. 1986a. Induction of leaf red-speckling by phosphorus on sorghum cultivars grown under controlled conditions. Crop Sci. 26: 551-557.

Fulrani, A.M.C., R.B. Clark, C.Y. Sullivan and J.W. Maranville. 1986b. Sorghum genotype differences to leaf "red-speckling" induced by phosphorus. J. Plant Nutr. 9: 1435-1451.

Gabor, T.S., H.R. Murkin, M.P. Stainton, J.A. Boughen and R.D. Titman. 1994. Nutrient additions to wetlands in the Interlake region of Manitoba, Canada: effects of a single pulse addition in spring. Hydrobiol. 279/280: 497-510.

Gagne F., J. Ridal, C. Blaise and B. Brownlee 1999. Toxicological effects of geosmin and 2-methylisoborneol on rainbow trout hepatocytes. Bull. Environl Contam Toxicol. 63: 174-180.

Gagnon, B., R. Robitaille and R.R. Simard. 1999. Characterization of several on-farm and industrial composted materials. Can. J. Soil Sci. 79: 201-210.

Galloway, J.N., W.H. Schlesinger, H. Levy II, A. Michaels and J.L. Schnoor. 1995. Nitrogen fixation: anthropogenic enhancement - environmental response. Global Biogeochem. Cycles. 9: 235-252.

Garbuz, A.M. 1971. Changes in the body's functional conditions in symptomless methaemoglobinaemia. *In* Methaemoglobineamia caused by various factors and methods of preventing it. Summaries of papers read at the First Scientific Conference on Methaemoglobinaemia. Leningrad, Russia. p. 18-21.

Gartner Lee Ltd. 1997. Fraser Basin Landfill Inventory. Prepared for the Fraser River Action Plan, Environment Canada. DOE FRAP Report No. 1997-19.

Gaudreau, D. and M. Mercier. 1997. La contamination de l'eau des puits privés par les nitrates en milieu rural. Direction de la Santé Publique de la Montérégie. 75 p.

George, C. 1995. Status: International wildland fire foam specification and NFPA 298- foam chemicals for fire control. National Wildlife Suppression Technology Program, Washington D.C. 26 p.

GESAMP- IMO/FAO.UNESCO/WMO/WHO/AEA/UN/UNEP. Joint group of experts on the scientific aspects of marine pollution. 1990. Review of potentially harmful substances. Nutrients. Rep. Stud. GESAMP. No. 34. Paris, France. 40 p.

Giddings, Michèle, Abdel-ilah Sadiki, Joanne Sketchell and Tim Macaulay. 2000. Cyanobacterial toxins: The development and evaluation of methods to determine microcystin levels in Canadian water supplies. *In* CWWA 9th National Drinking Water Conference, May 16-18, 2000, Regina, SK.

Gillı, G., G. Corrao and S. Favilli. 1984. Concentrations of nitrates in drinking water and incidence of gastric carcinomas: First descriptive study of the Piermonte Region, Italy. Sci. Total Environ. 34: 35-51.

Gilliam, F.S., M.B. Adams and B.M Yurish. 1996. Ecosystem nutrient responses to chronic nitorgen inputs at Fernow Experiment Forest, West Virginia. Can. J. For. Res. 26: 196-205.

Giroux, I. 1995. Contamination de l'eau souterraine par les pesticides et les nitrates dans les régions de culture de pommes de terre, Campagnes d'échantillonnage 1991-1992-1993. Ministère de l'Environnement et de la Faune, Québec, 61 p.

Glowka, L., F. Burhenne-Guilmin and H. Synge, in collaboration with J.A. McNeely and L Gündling. 1994. A guide to the convention on biological diversity. A contribution to the global biodiversity strategy. Environmental Policy and Law Paper No. 30. IUCN Environmental Law Centre and IUCN Biodiversity Programme. Gland, Switzerland, and Cambridge, U.K: IUCN (the World Conservation Union).

Gleig, D.B. and K.B. MacDonald. 1998. Nitrogen use efficiency in Canadian agriculture: a literature review. Final Draft. Written for the Canadian Fertilizer Institute, Ottawa, ON.

Glooschenko, W.A. 1979. Coastal salt marshes in Canada. *In* Proceedings of the Workshop on Canadian Wetlands. *Edited by* C.D.A. Rubec and F.C. Pollett. Lands Directorate, Environment Canada, Ecological Land Classification series. No. 12. Ottawa, ON. p. 69-76.

Goluboff, N. 1948. Case reports: Methaemoglobinaemia in an infant. J. Can. Med. Assoc. 58: 601-603.

Gomolka, R.E. 1975. An investigation of atmospheric phosphorus as a source of a lake nutrient. M.Sc. Thesis. University of Toronto, Toronto, ON. 167 p.

Goolsby, D.A., R.H. Coupe, D.J. Markovchick. 1991. Distribution of selected herbicides and nitrate in the Mississippi River and its major tributaries: U.S. Geological Survey Water-Resources Investigation. Rpt. No. 91-4163.

Gorham, E., S.E. Bayley and D.W. Schindler. 1984. Ecological effects of acid deposition upon peatlands: A neglected field in acid rain research. Can. J. Fish. Aquat. Sci 41: 1256-1268.

Gorham., P.R. 1964. Toxic algae. *In* Algae and Man. *Edited by* D.F. Jackson.Plenum Press. N.Y. p. 307-336.

Gorham, P.R. and W.W. Carmichael. 1980. Toxic substances from freshwater algae. Prog. Water Tech. 12: 189-198.

Goss, M.J., D.A.J. Barry and D.L. Rudolph. 1998. Contamination in Ontario farmstead domestic wells and its association with agriculture: 1. Results from drinking water wells. J. Contam. Hydrol. 32: 267-293.

Government of Canada. 1991. The state of Canada's environment. Ottawa: Environment Canada, State of the Environment Reporting, Ottawa, ON.

Government of Canada. 1995. CEPA review: the government Response. Environmental Protection Legislation Designed for the Future- A Renewed CEPA. Response to the Recommendations of the Standing Committee on Environment and Sustainable Development outlined in its fifth Report: It's about our Health! Towards Pollution Prevention. Ministry of Supply and Services Canada. Ottawa, ON. p. 82.

Government of Canada. 1996. The state of Canada's environment –1996. Environment Canada, State of the Environment Reporting Ottawa, ON.

Graham, D.M., J.A. Dahl, E.S. Millard, O.E. Johannsson and L.L. White. 1996. Assessment of the abundance, biomass and production of the lower trophic levels in the eastern basin of Lake Erie, 1994. Can. Tech. Rep. Fish. Aquat. Sci. No. 2110. 104 p.

Graham, M.D. 1997. Omnivory and selective feeding by zooplankton along a lake production gradient: Complementary 15N isotope and gut pigment analyses. M.Sc. thesis, Biology Department, University of Regina, Regina, SK.

Granéli, E., K. Wallström, U. Larsson, W. Granéli and R. Elmgren. 1990. Nutrient limitation of primary productivity in the Baltic Seas area. Ambio 19: 142-151.

Grant, G.A. and E.O. Hughes. 1953. Development of toxicity in blue-green algae. Can. J. Pub. Hlth. 44: 334-339.

Grennfelt, P. and H. Hultberg. 1986. Effects of nitrogen deposition on the acidification of terrestrial and aquatic ecosystems. Water Air Soil Pollut. 30: 945-963.

Grenon, S. 1994. The Impact of phosphates released from automatic dishwashing detergents. Environment Canada, Toxic Substances and Air Pollution Control, Québec Region, Montreal, QC.

Greven, H.C. 1992. Changes in the moss flora of the Netherlands. Biol. Conser. 59: 133-137.

Griffiths, D.E. and G.H. Townsend. 1987. Northern deltas- oases for wildlife. *In* Proceedings from the 1987 Symposium on Wetlands/Peatlands. *Edited by* C.D.A. Rubec and R.P. Overend. Edmonton. AB. Aug. 23-27. p. 551-557.

Grime, J.P. 1973. Control of species density in herbaceous vegetation. J. Environ. Mgmt. 1: 151-167.

Grime, J.P. 1977. Evidence for the existence of three primary strategies in plants and its relevance to ecology and evolutionary theory. Amer. Nat. 3: 1169-1194.

Grout, J.A., C.D. Levings and J.S. Richardson. 1997. Decomposition rates of purple loosestrife (*Lythrum salicaria*) and Lyngbyei's sedge (*Carex lyngbyei*) in the Fraser River Estuary. Estuaries 20: 96-102.

Grundon, N.J., D.J. Edwards, P.N. Takkar, C.J. Asher and R.B. Clark. 1987. Nutritional disorder of grain sorghum. Australian Centre of International Agricultural Research (ACIAR), Indian Council of Agricultural Research and University of Queensland. ACIAR monograph No. 2. 99 p.Halifax Regional Municipality. 1998. Halifax Regional Municipality Home Page. <www.region.halifax.ns.ca> (10 Dec. 1998).

Hall, J.P. 1995. Forest health monitoring in Canada: How healthy is the boreal forest? Water, Air and Soil Pollut. 82: 77-85.

Hall, K.J. and I. Yesaki. 1987. Primary production, detritus flux and nutrient cycling in tidal bulrush marshes of the Fraser River estuary. *In* Proceedings from the 1987 Symposium on Wetlands/Peatlands. *Edited by* C.D.A. Rubec and R.P. Overend Edmonton. AB. Aug. 23-27. p. 573-580.

Hall, P., W. Bowers, H. Hirvonen, G. Hogan, N. Forster, I. Morrison, K. Percy, R. Cox and P. Arp. 1997. Canadian Acid Rain Assessment. Volume 4: The effects on Canada's forests. Supply and Services Canada, Ottawa, ON. 47 p.

Hall, P.O.J., O. Holby, S. Kollberg and M-O. Samuelsson. 1992. Chemical fluxes and mass balances in a marine fish cage farm. IV. Nitrogen. Mar. Ecol. Prog. Ser. 89: 81-91.

Hall, R.I., P.R. Leavitt, R. Quinlan, A.S. Dixit and J.P. Smol. 1999. Effects of agriculture, urbanization and climate on water quality in the northern Great Plains. Limnol. Oceanogr. 44: 739-756.

Hallegraeff, G.M. 1993. A review of harmful algal blooms and their apparent global increase. Phycol. 32: 79-99.

Hallegraeff, G., D. Anderson and A. Cembella (eds). 1995. Manual on Harmful Marine Algae. IOC Manual No. 33. UNESCO.

Hamilton, S.J., S.F. McDonald, M. Galkowski and K.J. Buhl. 1994. Toxicity of fire retardant chemicals to aquatic organisms. *In* Foam Applications for Wildland and Urban Fire Management. *Edited by* National Wildfire co-ordinating group and Canadian committee on forest fire management. International wildland fire foam symposium and workshop, Thunder Bay, ON. May 3-7 1994. 17 p.

Hansen, L., J. Kristiansen and J. Rasmussen. 1994. Potential toxicity of the freshwater *Chrysochromulina* species *C. parva* (Prymnesiophyceae). Hydrobiol. 287: 157-59.

Harader Jr., R.R. and G.H. Allen. 1983. Ammonia toxicity to Chinook salmon parr: Reduction in saline water. Trans. Amer. Fish. Soc. 112: 834-837.

Hargrave, B.T., D.E. Duplisea, E. Pfeiffer and D.J. Wildish. 1993. Seasonal changes in benthic fluxes of dissolved oxygen and ammonium associated with marine cultured Atlantic salmon. Mar. Ecol. Prog. Ser. 96: 249-257.

Harman, J., W.D. Robertson, J.A. Cherry and L. Zanini. 1996. Impacts on a sand aquifer from an old septic system: nitrate and phosphate. Ground Water. 34: 1105-1114.

Harrison, P.J. 1985. Monitoring nutrients and plankton in the vicinity of the Iona Island Sewage outfall. Environment Canada. EPS. Pacific Region. # 85-04 59 p.

Harrison, P.J., J.D. Fulton, F.J.R. Taylor and T.R. Parsons. 1983. Review of the biological oceanography of the Strait of Georgia: pelagic environment. Can. J. Fish. Aquat. Sci. 40: 1064-1094.

Harrison, P.J., D.L. Mackas, B.W. Frost, R.W. Macdonald and E.A. Crecelius. 1994. An assessment of nutrients, plankton and some pollutants in the water column of Juan de Fuca Strait, Strait of Georgia and Puget Sound and their transboundary transport. *In* Review of the Marine environment and biota of Strait of Georgia, Puget Sound and Juan de Fuca Strait: Proceedings of the BC/Washington Symposium on the Marine Environment, Jan. 13-14, 1994. *Edited by* R.C.H. Wilson, R.J. Beamish, F. Aitkens and J. Bell. Can. Tech. Rep. Fish. Aquat. Sci. No. 1948. p. 138-174.

Hartman, P.E. 1983. Review: putative mutagens and carcinogens in foods. 1. Nitrate/nitrite ingestion and gastric cancer mortality. Environ. Mutagen. 5: 111-126.

Hatcher, A.I. and D.G. Patriquin. 1981 Salt marshes in Nova Scotia. A Status Report of the Salt Marsh Working Group. Dalhousie University, Halifax, NS. 70 p.

Hazel, C.R., W. Thomsen and S.J. Meith. 1971. Sensitivity of striped bass and stickleback to ammonia in relation to temperature and salinity. Cal. Fish Game 57: 138-153.

Hazel, R.H., C.E. Burkhead and D.G. Huggins. 1979. The development of water quality criteria for ammonia and total residual chlorine for the protection of aquatic life in two Johnson County, Kansas, streams. U.S. Department of the Interior, Office of Water Res. and Tech., Washington, D.C.

HCSCESD [House of Commons Standing Committee on Environment and Sustainable Development]. 1995. It's about our health! Towards pollution prevention. CEPA revisited. 5th report. 357 p.

Health and Welfare Canada. 1983. Guidelines for Canadian recreational water quality. Federal-Provincial Advisory Committee on Environment and Occupational Health. Ottawa, ON.

Health Canada. 1992. Nitrate/Nitrite. Guidelines for Canadian drinking water quality supporting document June 1987 (edited October 1992). Ottawa, ON. Unpublished. 8 p. (Note: all supporting dcouments are available in print copy either through Drinkng Water Sub-committee (Health Canada is the Secreatriat) or at www.hc-sc.gc.ca/waterquality (/eauqualite)).

Health Canada. 1993. Guidelines for Canadian Drinking Water Quality -- Supporting Documentation for Trihalomethanes. Ottawa, ON.

Health Canada. 1996. Guidelines for Canadian drinking water quality. Sixth Edition. Ottawa, ON. 90 p.

Health Canada. 1998a. Cyanobacterial toxins - Microcystin in drinking water. Document for public comment. Federal-Provincial Subcommittee on Drinking Water, Ottawa, ON. 26 p.

Health Canada. 1998b. Blue-Green Algae (cyanobacteria) and their Toxins. It's Your Health Fact Sheet. Ottawa, ON. 4 p.

Heatherington, E. 1985. Streamflow nitrogen loss following forest fertilization in a southern Vancouver Island watershed. Can. J. For. Res. 15: 34-41.

Hecnar, S.J. 1995. Acute and chronic toxicity of ammonium nitrate fertilizer to amphibians from southern Ontario. Environ. Toxicol. Chem. 14: 2131-2137.

Hecnar, S.J. and R.T. M'Closkey. 1996. Regional dynamics and the status of amphibians. Ecology. 77 (7): 2091-2097.

Heinze, R. 1999. Toxicity of the cyanobacterial toxin microcystin-LR to rats after 28 days intake with the drinking water. Environ. Toxicol. 14: 57-60.

Hendry, M.J., R.W. Gillham and J.A. Cherry. 1983. An integrated approach to hydrogeological investigations: A case history. J. Hydrol. 63: 211-232.

Hendry, M. J., R. G. L. McCready and W. D. Gould. 1984. Distribution, source and evolution of nitrate in a glacial till of southern Alberta, Canada. J. Hydrol. 70: 177-198.

Henry, J. L. and W. A. Meneley. 1993. Fertilizers and Groundwater Nitrate. Western Canada Fertilizer Association, Surrey, BC.

Hess, P.J. 1986. Ground-Water Use in Canada, 1981. NHRI Paper No. 28, IWD Technical Bulletin No. 140, Environment Canada, Ottawa, ON.

Hibbs, C.M., E.L. Stencel and R.M. Hill. 1978. Nitrate toxicosis in cattle. Vet. Hum. Toxicol. 20: 1-2.

Hickey, C.W. and M.L. Vickers. 1993. Toxicity of ammonia to nine native New Zealand freshwater invertebrate species. Arch. Environ. Contam. Toxicol. 26: 292-298.

Hill, A.C. and J.H. Bennet. 1970. Inhibition of apparent photosynthesis by nitrogen oxides. Atmos. Environ. 4: 341-348.

Hill, M.J., G.M. Hawksworth and G. Tattersall. 1973. Bacteria, nitrosamines and cancer of the stomach. Br. J. Cancer. 28: 562-574.

Hodson, P.V., U. Borgmann and H. Shear. 1979. Toxicity of copper to aquatic biota. *In* Copper in the Environment. *Edited by* J.O. Nriagu. John Wiley and Sons. New York. p. 307-372.

Holby, O. and P.O.J. Hall. 1991. Chemical fluxes and mass balances in a marine fish cage farm. II. Phosphorus. Mar. Ecol. Prog. Ser. 70: 263-272.

Holt, G.J. and C.R. Arnold 1983. Effects of ammonia and nitrite on growth and survival of red drum eggs and larvae. Trans. Amer. Fish. Soc. 112: 314-318.

Holtan, H., L. Kamp-Nielsen and A.L. Stuanes. 1988. Phosphorus in soil, water and sediment: an overview. Hydrobiol. 170: 19-34.

Hook, D. 1993. Wetlands: History, current status, and future trends. Environ. Toxicol. Chem. 12: 2157-2166.

Hornbeck, J., C.W. Martin, R. S. Pierce, F.H. Bormann, G.E. Likens and J.S. Eaton. 1987. The northern hardwood forest ecosystem: Ten years of recovery from clearcutting. USDA For. Serv. Res. Paper. NE-RP-596, Northeastern Forest Experiment Station, Bloomall, Pennsylvania.

Houle D., R. Paquin, C. Camire, R. Ouimet and I. Duchesne. 1997. Response of the Lake Clair Watershed (Ducsesnay, Quebec) to changes in precipitation chemistry (1988-1994). Can. J. For. Res. 27: 1813-1821.

Howarth, R.W. 1988. Nutrient limitation of net primary production in marine ecosystems. Ann. Rev. Ecol. 19: 89-110.

Howarth, RW. 1993. The role of nutrients in coastal waters (Appendix A). *In* Managing Wastewater in Coastal Urban Areas. *Edited by* J.J. Boland, B.P. Anderson, N.H. Brooks, W.M. Eichbaum, L.R. Goldman, D.R.G. Harleman, R.W. Howarth, R.J. Hugget, T.M. Keinath, A.J. Mearns, C.R. O'Melia, L.A. Roesner, J.B. Rose and J.R. Schubel. National Academy Press, Washington, DC. p. 177-202.

Howarth, R.W., G. Billen, D. Swaney, A. Townsend, N. Jaworski, K. Lajtha, J.A. Downing, R. Elmgren, N. Caraco, T. Jordan, F. Berendse, J. Freney, V. Kudeyarov, P. Murdoch and Z. Zhao-liang. 1996. Regional nitrogen budgets and riverine inputs of N and P for the drainages to the North Atlantic Ocean: natural and human influences. Biogeochem. 35: 75-139.

Howell, E.T., Marvin, C.H., Bilyea, R.W., Kauss, P.B. and K. Sommers. 1996. Changes in environmental conditions during *Dreissena* colonization of a monitoring station in eastern Lake Erie. J. Great Lakes Res. 22(3):744-756.

Howes, B.L., J.W.H. Dacey and D.D. Goehringer. 1986. Factors controlling the growth form of *Spartina alterniflora*: Feedbacks between above ground production, sediment oxidation, nitrogen, and salinity. J. Ecol. 74: 881-898.

Hoyer, M.V. and D.E. Canfield Jr. 1991. A phosphorus-fish standing crop relationship for streams? Lake Res. Manage. 7:25-32.

Huddart, P.A., F.J. Longstaffe and A.S. Crowe. 1999. δD and $\delta\ ^{18}$O evidence for inputs to groundwater at a wetland coastal boundary in the southern Great Lakes region of Canada. J. Hydrol. 214: 18-31

Hudon, C. and A. Sylvestre. 1998. Qualité de l'eau en aval de l'archipel de Montréal, 1994-1996. Environnement Canada – Région du Québec, Conservation de l'environnement, Centre Saint-Laurent. Rapport scientifique et technique ST-170, 338 p.

Hudson, R.J.M., S.A. Gherini and R.A. Goldstein. 1994. Modeling the global carbon cycle: nitrogen fertilization of the terrestrial biosphere and the "missing" CO_2 sink. Glogal Geochem. Cyc. 8: 307-333.

Huey , D.W. and T.L Beitinger. 1980a. Hematological responses of larval *Rana catesbiana* to sublethal nitrite exposures. Bull. Environ. Contam. Toxicol. 25:574-577.

Huey, D.W. and T.L. Beitinger. 1980b. Toxicity of nitrite to larvae of the salamander *Ambystoma texanum*. Bull. Environ. Contam. Toxicol. 25: 909-912.

Hutchinson, I. 1982. Vegetational-environmental relationship in a brackish marsh, Lula Island, British Columbia, Canada. Can. J. Bot. 60: 452-462.

Idler, D.R., G.L. Fletcher and R.F. Addison. 1981. Effects of yellow phosphorus in the Canadian environment. National Research Council Canada. Ottawa, ON. 110 p.

Ingram, W. and G.W. Prescott. 1954. Toxic freshwater algae. Amer. Mid. Nat. 52: 75-87.

IRC [Integrated Resource Consultants Inc.]. 1994a. Agricultural Land Use Survey in the Sumas River Watershed: Summary Report. Environment Canada. Conservation and protection. Vancouver, BC. FRAP # 1994-21. 112 p.

IRC [Integrated Resource Consultants Inc.]. 1994b. Wastewater Characterization of Four Industrial Discharges in the Fraser River Basin. Volume 1. Environment Canada. Vancouver, BC. DOE FRAP 1994-09. 72 p.

Isom, B.G. 1960. Toxicity of elemental phosphorus. J. Water Pollut. Contr. Fed. 32: 1312-1316.

Jacobs, D. 1999. How farmers are killing frogs: Fertilizer runoff is flowing freely into ponds, rivers, and the water table. The Ottawa Citizen. Ottawa, ON. Tuesday March 9, 1999. p. A5.

Jacobson, J.S., T. Bethard, L.I. Heller and J.P. Lassoie. 1990. Respiration of *Picea rubens* seedlings to intermittent mist varying in acidity and in concentrations of sulphur and nitrogen containing pollutants. Physiol. Plant. 78: 595-601.

Jangaard, P.M. 1972. Effects of elemental phosphorus on marine life: Collected papers resulting from the 1969 pollution crisis, Placentia Bay, NF. 313 p.

Janus, L.L. and R.A Vollenweider. 1981. The OECD co-operative programme on eutrophication. Canadian contribution. Summary report. National Water Research Institute Inland Waters Directorate. Scientific Series No. 131. Burlington, ON. 380 p.

Janzen, H.H., R.L. Desjardins, J.M.R. Asselin and B. Grace. 1998. The health of our air. Agriculture and Agri-Food Canada, Ottawa, ON. 98 p.

Jefferies, R.L. and N. Perkins. 1977. The effects on the vegetation of additions of inorganic nutrients on salt marsh soils at Stiffkey, Norfolk. J. Ecol. 65: 867-882.

Jeffries, D.S. 1995. A preliminary assessment of nitrogen-based fresh water acidificaton in southeastern Canada. Water Air Soil Pollut. 85: 433-438.

Jeffries, D.S., W.R. Snyder, W.A. Scheider and M. Kirby. 1978. Small-scale variations in precipitation loading near Dorset, Ontario. Water Pollut. Res. J. Can. 13: 73-81.

Jensen, J.B. 1991. Environmental regulation of fresh water fish farms in Denmark. *In* Nutritional Strategies and Aquaculture Waste. Proceedings of the First International Symposium on Nutritional Strategies in Management of Aquaculture Waste (NSMAW). *Edited by* C.B. Cowey and C.Y. Cho. University of Guelph, Guelph, ON, 1990.

Jensen, S., C. Anders, L. Goatcher, T. Perley, S. Kenefick and S. Hrudey. 1994. Actinomycetes as a factor in odour problems affecting drinking water from the North Saskatchewan River. Water Res. 28: 1393-1401.

Jewett, K., D. Daugharty, H.H. Krause and P.A. Arp. 1995. Watershed responses to clear-cutting: Effects on soil solutions and stream water discharge in central New Brunswick. Can. J. Soil Sci. 75: 475-490.

Jochimsen, E.M., W.W. Carmichael, J. An, D.M. Cardo, S.T. Cookson, C.E.M. Holmes, M.B. de C Antunes, D.A.de Melo Filho, T.M. Lyra, V.S.T. Barreto, S.M.F.O. Azevedo and W.R. Jarvis. 1998. Liver failure and death after exposure to microcystins at a hemodialysis center in Brazil. N. Engl. J. Med. 338: 873-878.

Johnsen, F. and A. Wandsvik. 1991. The impact of high energy diets on pollution control in the fish farming industry. *In* Nutritional Strategies and Aquaculture Waste. *Edited by* C.B. Cowey and C.Y. Cho. Proceedings of the First International Symposium on Nutritional Strategies in Management of Aquaculture Waste (NSMAW). University of Guelph, Guelph, ON, 1990.

Johnson, J.L., N.R. Schneider, C.L. Kelling and A.R. Koster. 1983. Nitrate exposure in prenatal beef calves. *In* Proceedings of Aver. Assn. Vet. Lab. Diag. 26th annual meeting. p. 14-32.

Johnson, W.W. and H.O. Sanders. 1977. Chemical forest fire retardants: Acute toxicity to five freshwater fishes and a scud. Technical papers of the U.S. Fish and Wildlife Service. No. 19. U.S. Department of the Interior, Washington, DC.

Johnston, C.T., P.G. Cook, S.K. Frape, L.N. Plummer, E. Busenberg and R.J. Blackport. 1998. Ground water age and nitrate distribution within a glacial aquifer beneath a thick unsaturated zone. Ground Water. 36: 171-180.

Johnston, N.T., C.J. Perrin, P.A. Slaney and B.R. Ward. 1990. Increased juvenile salmonid growth by whole-river fertilization. Can. J. Fish. Aquat. Sci. 47: 862-872.

Jones, G., S. Gurney and D. Rocan. 1998. Blue-green algae and microcystin-LR in surface water supplies of southwestern Manitoba. Manitoba Environment. Winnipeg, MB. Report No. 98-06. 82 p.

Jones, G.J. and P.T. Orr. 1994. Release and degradation of microcystin following algaecide treatment of a *Microcystis aeruginosa* bloom in a recreational lake, as determined by HPLC and protein phosphatase inhibition assay. Water Res. 28: 871-876.

Jones, J.B. 1988. Phosphorus toxicity in tomato plants: When and how does it occur?Commun. Soil Sci. Plant Anal. 29 (11-14): 1779-1784

Jüttner, F. 1995. Physiology and biochemistry of odorous compounds from freshwater cyanobacteria and algae. Wat. Sci. Technol. 31:69-78

Jüttner, F. 1999. Allelochemical control of natural photoautotrophic biofilms. Royal Soc. of Chemistry, special publications 242: 43-50.

Kadlec, R.H. and R.L. Knight. 1996. Treatment Wetlands. Lewis Publishers. Boca Raton, FL, U.S.A. 894 p.

Kahl, J.S., S.A. Norton, I.J. Fernandez, K.J. Nadelhoffer, C.T. Driscoll and J.D. Aber. 1993. Experimental inducement of nitrogen saturation at the watershed scale. Environ. Sci. Tech. 27: 565-568.

Kaiser, J. 1999. A trematode parasite causes some frog deformities. Science. 284: 731-732.

Kamiya, H., Naka, K., and Hashimoto, K. 1979. Ichthyotoxicity of a flagellate *Uroglena volvox*. Bull. Jap.Soc. Sci. Fish. 45:129.

Kamstra, A., J.A. Span and J.H. vanWeerd. 1996. The acute toxicity of sublethal effects of nitrite on growth and feed utilization of European eel, *Anguilla anguilla (L.)*. Aqua. Res. 27: 903-911.

Kay, R.M. 1949. Limnological studies of the Dundas Marsh Region. MA. Thesis. McMaster University, Hamilton, ON.

Keddy, C. 1988. A review of *Lythrum salicaria* (purple loosestrife) ecology and management: the urgency for management in Ontario (Toronto: Natural Heritage League).

Keeney, D.R. 1987. Sources of nitrate to groundwater. CRC Crit. Rev. Environ. Contam. 10: 257-304.

Keeney, D.R. and T.H. Deluca. 1993. Des Moines River nitrate in relation to watershed agricultural practices: 1945 versus 1980s. J. Environ. Qual. 22: 267-272.

Keizer, P.D. and D.C. Gordon Jr. 1985. Nutrient dynamics in Cumberland Basin-Chignecto Bay, a turbid macrotidal estuary in the Bay of Fundy, Canada. Neth. J. Sea Res. 19 (3/4): 193-205.

Kenefick, S., S. Hrudley, E. Prepas, N. Motkosky and H. Peterson. 1992. Odorous substances and cyanobacterial toxins in prairie drinking water sources. Wat. Sci. Technol. 25: 147-154.

Kenney, B.C. 1990. Dynamics of phosphorus in a chain of lakes: The Fishing Lakes. N.H.R.I. Paper No. 44. I.W.D. Scientific Series No. 176.

Kern-Hansen, U. and F.H. Dawson. 1978. The standing crop of aquatic plants of lowland streams in Denmark and the inter-relationships of nutrients in plant, sediment and water. Proceedings of the European Weed Research Society 5th Symposium on Aquatic Weeds. p. 143-150.

Kerr, M.A. 1987. Land use pressures on British Columbia's coastal wetlands. *In* Proceedings from the 1987 Symposium on Wetlands/Peatlands. *Edited by* C.D.A. Rubec and R.P. Overend. Edmonton. Alberta. Aug. 23-27. p. 349-354.

Kerr, S.R. and R.A. Ryder. 1992. Effects of cultural eutrophication on coastal marine fisheries: a comparative approach. Sci. Total Environ. (Suppl.): 599-614.

Kettrup, A. and U. Hüppe. 1988. Phosphorus. *In* Handbook on Toxicity of Inorganic Compounds. *Edited by* H.G. Seiler and H. Sigel. Marcel Dekker, Inc., New York. p. 521-532.

Kim Y, Y. Lee, C. Gee and E. Choi. 1997. Treatment of taste and odor causing substances in drinking water. Wat. Sci. Technol. 35:29-36

Kincheloe, J.W., G.A. Wederneyer and D.L. Koch. 1979. Tolerance of developing salmonid eggs and fry to nitrate exposure. Bull. Environ. Contam. Toxicol. 23: 574-578.

Kleijn, D. and M. Verbeek. 2000. Factors affecting the species composition of arable field boundary vegetation. J. Appl. Ecol. 37: 256-266.

Kleinjans, J.C.S., H.J. Albering, A. Marx, J.M.S. van Maanen, B. van Agen, F. ten Hoor, G.M.H. Swaen and P.L.J.M. Mertens. 1991. Nitrate contamination of drinking water: evaluation of genotoxic risk in human populations. Environ. Hlth. Perspect. 94: 189-200.

Koerselman, W. and J.T.A. Verhoeven. 1992. Nutrient dynamics in mires of various trophic status: nutrient inputs and outputs and the internal nutrient cycle. *In* Fens and Bogs in The Netherlands: Vegetation, History, Nutrient Dynamics and Conservation. *Edited by* J.T.A. Verheoven. Kluwer Publishing. Dordrecht, The Netherlands. p. 397-432.

Korom, S. F. 1992. Natural denitrification in the saturated zone: A review. Water Resour. Res. 28: 1657-1668.

Kotak, B.G., S.L. Kenefick, D.L. Fritz, C.G. Rousseaux, E.E. Prepas and S.E. Hrudey. 1993a. Occurrence and toxicological evaluation of cyanobacterial toxins in Alberta Lakes and farm dugouts. Water Res. 27: 495-506.

Kotak, B.G., S.E. Hrudey, S. Kenefick and E.E. Prepas. 1993b. Toxicity of cyanobacterial blooms in Alberta Lakes. *In* Proceedings of the Nineteenth Annual Aquatic Toxicity Workshop. *Edited by* E.G. Baddaloo, S. Ranamoorthy and J.W. Moore. Can. J. Fish. Aquat. Sci. Tech. Rep. No. 1942. p. 172-179.

Kotak, B., R. Zurawell, E. Prepas and C. Holmes. 1994. Microcystin-LR concentration in aquatic food web compartments from lakes of varying trophic status. Can. J. Fish. Aquat. Sci 53: 1974-1985.

Kotak, B.G., A.K-Y. Lam, E.E. Prepas, S.L. Kenefick and S.E. Hrudey. 1995. Variability of the hepatotoxin microcystin-LR in hypereutrophic drinking water lakes. J. Phycol. 31: 248-263.

Krause, H.H. 1982. Nitrate formation and movement before and after clear-cutting of a monitored watershed in central New Brunswick. Can. J. For. Res. 12: 922-930.

Kvasnicka, B. and L.J. Krysl. 1996. Nitrate poisoning. *In* Nitrate poisoning in livestock. Cattle producer's library CL 620. University of Nevada. Las Vegas, NV. <www.forages.css.orst.edu/Topics/Pastures/Species/Grasses/Animal_issues/Nitrate.html>.

Laak, R. 1974. Relative pollution strengths of undiluted waste materials discharged in households and the dilution waters used for each. *In* Manual of Grey Water Treatment Practice. *Edited by* J. H. T. Winneberger. Ann Arbor Science, Ann Arbor, MI. p. 68-78.

Labat-Anderson Inc. 1996. Chemicals used in wildland fire suppression: A risk assessment. Arlington, VA. 139 p. + Appendices.

Lambert, T.W., C.F.B. Holmes and S.E. Hrudey. 1994. Microcystin class of toxins: health effects and safety of drinking water supplies. Environ. Rev. 2: 167-186.

Landau, M. 1992. Introduction to aquaculture. John Wiley and Sons, Inc., New York. 440 p.

Lange, C.L. and S.A. Wittmeyer. 1997. The contribution of zebra mussel (*Dreissena spp.*) feces and pseudofeces production to taste and odor episodes in the Niagara River and Lake Erie. *In* Zebra Mussels and Aquatic Nuisance Species. *Edited by* F.M D'Itri. Ann Arbor Press, Chelsea, MI. p. 225-243.

Larney, F.J. 1992. Disposing of farm waste in an environmentally friendly manner – A management headache of the '90's. *In* Advances in dairy technology. Proceedings of the 1992 Western Canadian Dairy Seminar. *Edited by* J. Kennelly. Red Deer, AB March 11-13, 1992. p. 137-150.

Lamontagne, S., R. Carignag, P. D'Arcy, Y.T. Prairie and D. Paré. 2000. Element export in runoff from eastern Canadian Boreal Shield drainage basins following forest harvesting and wildfires. Can. J. Fish. Aquat. Sci. 57 (Suppl. 2): 118-128.

Leach, B.A. 1982. Waterfowl on a Pacific flyway. Special Pub. No. 5, B.C. Prov. Museum, Victoria, BC. 211 p.

Lee, J.A., J.A. Badderley and S.R. Woodin. 1989. Effects of acidic deposition of semi-natural vegetation. *In* Acidification in Scotland. Edinburgh Scottish Development Department.

Leeper, D and Porter, K. 1995. Toxicity of the mixotrophic chrysophyte *Poteriochromonas malhamensis* to the cladoceran *Daphnia ambigua*. Archiv. Hydrobiol. 134: 207-22.

Leonardi, S. and W. Flückiger. 1989. Effects of cation leaching on mineral cycling and transpiration: investigation with beech seedlings, *Fagus sylvatica* L. New Phytol. 111: 173-179.

Levings, C.D. 1994. Some ecological concerns for net-pen culture of salmon on the coasts of the northeast Pacific and Atlantic Oceans, with special reference to British Columbia. J. Appl. Aqua. 4: 65-141.

Lesht, B.M. T.D. Fontaine and D.M. Dolan. 1991. Great Lakes total phosphorus model: post and regionalized sensitivity analysis. J. Great Lakes Res. 17: 3-17.

Lester, P. 1998. Nitrate poisoning. Specialist Agriculture Supply Pty. Ltd. Consultant Services, Melbourne, Australia. <cpc.netconnect.com.au/~sas/index.html>.

Liebscher, H., B. Hii and D. McNaughton 1992. Nitrates and pesticides in the Abbotsford Aquifer, southwestern British Columbia. Environment Canada, Pacific and Yukon Region, Vancouver.

Linsey, G.A., D.W. Schindler and M.P. Stainton. 1987. Atmospheric deposition of nutrients and major ions at the Experimental Lakes Area in Northwestern Ontario, 1970 to 1982.

Litav, M. and Y. Lehrer 1978. The effects of ammonium in water on *Potamageton lucens*. Aquat. Biol. 5: 127-138.

Lohman, K. J.R. Jones and B.D. Perkins. 1992. Effects of nutrient enrichment and flood frequency on periphyton biomass in Northern Ozark streams. Can. J. Fish. Aquat. Sci. 49: 1198-1205.

Lougheed, V.L. and P. Chow-Fraser. 1998. Factors that regulate the zooplankton community structure of a turbid, hypereutrophic Great Lakes wetland. Can. J. Fish. Aquat. Sci. 55: 150-161.

Luca, D., V. Luca, F. Cotor and L. Rileanu. 1987. *In vivo* and *in vitro* cytogenetic damage induced by sodium nitrate. Mutat. Res. 189: 333-357.

Lucas, P.W. 1990. The effects of prior exposure to sulphur dioxide and nitrogen dioxide on the water relations of Timothy grass (*Phloem pratense*) under drought conditions. Environ. Pollut. 66: 117-138.

MacDonald, K.B. 1977. Plant and animal communities of Pacific North American salt marshes *In* Wet Coastal ecosystems. *Edited by* V.J. Chapman Elsevier, Amsterdam. p. 167-191.

MacDonald, K.B. 2000a. Residual Nitrogen. *In* Environmental sustainability of Canadian agriculture: Report of the agri-environmental indicator project. *Edited by* T. MacRae, C.A.S. Smith and L.J. Gregorich. Research Branch, Policy Branch, Prairie Farm Rehabilitation Administration, Agriculture and Agri-Food Canada.

MacDonald, K.B. 2000b. Risk of water contamination by nitrogen. *In* Environmental sustainability of Canadian agriculture: Report of the agri-environmental indicator project. *Edited by* T. MacRae, C.A.S. Smith and L.J. Gregorich. Research Branch, Policy Branch, Prairie Farm Rehabilitation Administration, Agriculture and Agri-Food Canada.

MacEwen, J.D. and E.H. Vernot. 1972. Toxic Hazards Research Unit annual technical report. NTIS AD. 358-758.

MacIsaac, E.A. and J.G. Stockner. 1995. The environmental effects of lakepen reared Atlantic salmon smolts. Report prepared for the Science Council of British Columbia. Dept. of Fisheries and Oceans, West Vancouver.

Mackay, D.M. 1990. Characterization of the distribution and behavior of organic contaminants in the subsurface. *In* Ground Water and Soil Contamination Remediation. National Academy Press, Washington, DC.

MacKenzie, D. 1998. Waste not. New Scientist (29 August 1998): 26-30.

MacKenzie, D.I. 1982. The dune-ridge forest, Delta Marsh, Manitoba: overstory vegetation and soil patterns. Can. Field Nat. 96: 61-68.

MacKinnon, K. and D.R. Scott. 1984. An evaluation of salt marshes in Atlantic Canada. Environment Canada. Lands and Integrated Programs Directorate. Atlantic Region. 42 p.

Mahendrappa, M.K., N.W. Foster, G.F. Weetman and H.H. Krause 1986. Nutrient cycling and availability in forest soils. Can. J. Soil Sci. 66: 547-572.

Mahmood, N.A. and W.W. Carmichael. 1987. Anatoxin *a* (s), an anticholinesterase from the cyanobacterium *Anabaena flos-aquae*. NRC-525-17. Toxicon. 25: 1221-1227.

Manitoba Agriculture. 1994. Farm practices guidelines for dairy (or poultry) (or beef) (or hog) producers in Manitoba.

Manitoba Agriculture. 1999. General recommendations for fertilizer. <www.gov.mb.ca/cgi-bin/print…mz01s00html?field+crop+fertilizer> (3 Feb. 1999).

Manitoba Department of Mines, Resources and Environmental Management, Environmental Management Division, Environmental Protection Branch, Water Quality Study, South Portion of Lake Winnipeg, 1973, Report No. 24, June 1974.

Marco, A. and A.R. Blaustein. 1999. The effects of nitrite on behaviour and metamorphosis in cascades frogs (*Rana cascadae*). Environ. Toxicol. Chem. 18: 946-949.

Markewitz, D. D.D. Richter, H.L. Allen and J.B. Urrego. 1998. Three decades of observed soil acidification in the Calhoun Experimental Forest: Has acid rain made a difference? Soil Sci. Soc. Amer. J. 62: 1428-1439.

Marsalek, J. 1984. Caracterisation du ruissellement de surface issu d'une zone urbaine commerciale. Sciences et techniques de l'eau. 17: 163-167.

Marsalek, J. and H.O. Schroeter. 1988. Annual loadings of toxic contaminants in urban runoff from the Canadian Great Lakes Basin. Water Pollut. Res. J. Can. 23: 360-378.

Marsalek, J. and S. Kok (Eds.). 1997. Great Lakes 2000 Cleanup fund program: stormwater management and control of combined sewer overflows. Water Qual. Res. J. Can. 32 (Theme issue): 1-226.

Martin, C.W., J. W. Hornbeck, G.E. Likens and D.C. Buso. 2000. Impacts of intensive harvesting on hydrology and nutrient dynamics of northern hardwood forests. Can. J. Fish. Aquat. Sci. 57 (Suppl. 2): 19-29.

Mason, C.F. 1991. Biology of freshwater pollution. Harlow: Longman Scientific and Technical. 250 p.

Mathias, M.S. and P. Moyle. 1992. Wetland and aquatic habitats. Agri. Ecosys. Environ. 42:156-176.

Matson, P.A., W.J. Parton, A.G. Power and M.J. Swift. 1997. Agricultural intensification and ecosystem properties. Science. 277: 504-509.

Maynard, D.N. and A.V. Barker. 1969. Studies on the tolerance of the plants to ammonium nutrition. J. Amer. Soc. Hortic. Sci. 94: 235-239.

McCormick, J., S. Broderius and J. Fiandt. 1984. Toxicity of ammonia to early life stages of the green sunfish, *Lepomis cyanellus*. Environ. Pollut. (Ser. A) 36: 147-163.

McCulloch, R.B., G.S. Few, G.C. Murray Jr. and V.P. Aneja. 1998. Analysis of ammonia, ammonium aerosols and acid gases in the atmosphere at a commercial hog farm in eastern North Carolina, USA. Environ. Pollut. 102 (S1): 263-268.

McGinn, S.M. and H.H. Janzen. 1998. Ammonia sources in agriculture and their measurement. Can. J. Soil. Sci. 78: 139-148.

McLarty, A and A.G. Thachuk. 1986. Cootes Paradise Study. Water Resources Assessment Unit. Technical Support Section, West Central Region, Hamilton, ON. Ministry of the Environment, unpublished technical report.

McNulty S.G., J.D. Aber and R.D. Boone. 1991. Spatial changes in forest floor and foliar chemistry of spruce-fir forests across New England. Biogeochem. 14: 13-29.

McNulty S.G., J.D. Aber and S.D. Newman. 1996. Nitrogen saturation in a high elevation spruce-fir stand. For. Ecol. Manage. 84: 109-121.

McLeod, J.A. and G.S. Bondar. 1952. A case of suspected algae poisoning in Manitoba. Can. J. Publ. Hlth. 43: 347-350.

MDMNR [Manitoba Department of Mines and Natural Resources]. 1968. The Delta Marsh: Its values, problems and potentialities. Winnipeg, MB. 75 p.

Meade, M.E. and S.A. Watts. 1995. Toxicity of ammonia, nitrite, and nitrate to juvenile Australian crayfish, *Cherax quadricarinatus*. J. Shell. Res. 14 (2): 341-346.

Medovy, H., W.C. Guest and M. Victor. 1947. Cyanosis in infants in rural areas. J. Can. Med. Assoc. 56: 505-508.

MEFQ [Ministère de l'Environnement et de la Faune du Québec]. 1990. Critères de qualité de l'eau. Service d'évaluation des rejets toxiques et direction de la qualité des cour d'eau. Ministère de l'Environnement du Québec. Québec. 423 p.

MEFQ [Ministère de l'Environnement et de la Faune du Québec]. 1995. Rapport d'évaluation des ouvrages municipaux d'assainissement des eaux du Programme d'assainissement des eaux du Québec (PAEQ) – Année 1993. Québec.

MEFQ [Ministère de l'Environnement et de la Faune du Québec]. 1997. Critères provisoires pour la valorisation des matières résiduelles fertilisantes.

MEFQ [Ministère de l'Environnement et de la Faune du Québec]. 1998a. Critères de qualité de l'eau de surface au Québec, Direction des écosytèmes aquatiques. Ministère de l'Environnement et de la Faune, Québec. 387 p.

MEFQ [Ministère d'Environnement et de la Faune du Québec]. 1998b. Regulation highlights: Reduction of pollution from agricultural sources. <www.mef.gouv.qu.ca/en/environn/agricultural/regulati.htm> (16 Dec. 1998).

Mehlhorn, H. and A.R. Wellburn. 1987. Stress ethylene formation determines plant sensitivity to ozone. Nature 327: 417-418.

MENV [Ministère de l'Environnement du Québec]. 2000. Bassin versant de la rivière Yamaska. <www.menv.gouv.qc.ca/sol/pratiques-agri/yamaska/intro.htm#occup> (1 Mar. 2000).

Miller, H.G., J.M. Cooper and O.J.L. Paulene. 1979. Nutrient cycles in pine and their adaptation to poor soils. Can. J. For. Res. 9: 19-26.

Mills C.J., J.B. Richard, K.P. Cantor, J. Reif, S.E. Hrudey, P. Huston and an Expert Working Group. 1998. Health Risks of Drinking Water Chlorination By-products: Report of an Expert Working Group. Chronic Diseases in Canada 19 (3).

Mills, K., S. Chalanchuk and D. Allan. 1995.The fish kills in Lake 302S, an experimentally acidified lake 1-261. Acid Reign'95, Göteborg, Sweden.

Mitchell, M.J., N.W. Foster, J.M. Shepard and I.K. Morrison. 1992. Nutrient cycling in Huntington Forest and Turkey Lakes deciduous stands:nitrogen and sulfur. Can. J. For. Res. 22: 457-464.

Mitchell, P. 1999. Water quality management in Lac Ste. Anne and Lake Isle: A diagnostic study. Water Sciences Branch, Alberta Environment, Edmonton, AB. 44 p.

Mitsch, W.J. 1992. Combining ecosystem and landscape approaches to Great Lakes wetlands. J. Great Lakes Res. 18: 552-570.

Mitsch, W.J and J.G. Gosselink. 1993. Wetlands, 2nd ed. Van Nostrand Reinhold, New York. 722 p.

Moccia, R.D. and D.J. Bevan. 1996. Aquastats 1995: Ontario aquacultural production in 1995 and situation outlook. Aquaculture Centre, University of Guelph. AEC Order No. 96-001. <www.aps.uoguelph.ca/~aquacentre/aec/publications/aquastats95/STATS95.PDF> (12 Dec. 2000).

Monsanto. 1991. Material safety data sheet: Phos-Chek Fire Retardant, grades D-75F and D-75R. St. Louis, MO.

Monteverde, C.A., R.L. Desjardins and E. Pattey. 1997. Estimates of nitrous oxide emissions from agroecosystems in Canada for 1986 and 1981 using the revised 1996 IPCC/OECD methodology. Agriculture and Agri-Food Canada Agri-Environmental Indicator Project, Agroecosystem Greenhouse Gas Balance Indicator: Nitrous Oxide Component. Technical Report No. 20.

Moore, D.R.J., P.A. Keddy, C.L. Gaudet and I.C. Wisheu. 1989. Conservation of wetlands: Do infertile wetlands deserve a higher priority? Biol. Conser. 47: 203-217.

Morgan, S.A., J.A. Lee and T.W. Ashenden. 1992. Effects of nitrogen oxides on nitrate assimilation in bryophytes. New Phytol. 120: 89-97.

Morris, J.T. 1991. Effects of nitrogen loading on wetland ecosystems with particular reference to atmospheric deposition. Ann. Rev. Ecol. Syst. 22: 257-279.

Morrison, I.K. and N.W. Foster 1977. Fate of urea fertilizer added to a boreal forest *Pinus banksiana* Lamb. stand. Soil Sci. Soc. Amer. J. 41: 441-448.

Mueller, D.K. and D.R. Helsel. 1996. Nutrients in the nation's waters- Too much of a good thing? U.S. Geological Survey Circular 1136. Denver, CO. 24 p.

Mulholland, P.J., A.D. Steinman and J.W. Elwood. 1990. Measurement of phosphorus uptake length in streams: comparison of radiotracer and stable PO_4 releases. Can. J. Fish. Aquat. Sci. 47: 2351-2357.

Mulligan, G.A. and D.B. Munro. 1990. Poisonous plants of Canada. Agriculture Canada Publication No. 1842/E. Ottawa, ON. 96 p.

Munroe. D.J. 1986. Qu'Appelle Fishing Lakes nutrient loading study, 1980 to 1983; Final report. WQB-WNR –86-02.

Murkin, E.J., H.R. Murkin and R.D. Titman. 1992. Nektonic invertebrate abundance and distribution at the emergent vegetation-open water interface in the Delta Marsh, Manitoba, Canada. Wetlands 12: 45-52.

Murkin, H.R. and B.D.J. Batt. 1987. Interactions of vertebrates and invertebrates in peatlands and marshes. Mem. Ent. Soc. Can. 140: 15-30.

Murkin, H.R., J.B. Pollard, M.P. Stainton, J.A. Boughen and R.D. Titman. 1994. Nutrient additions to wetlands in the Interlake region of Manitoba, Canada: effects of periodic additions throughout the growing season. Hydrobiol. 279/280: 483-495.

Murray, A.J.S. and A.R. Wellburn.1980. Differences in nitrogen metabolism between contuvars of tomato and pepper during exposure to glasshouse atmosphere containing oxides of nitrogen. Environ. Pollut 39: 303-316.

Nakajima, M., T. Ogura, Y. Kusama, N. Iwabuchi, T. Imawaka, A. Araki, T. Sasaki and E. Hirose. 1996. Inhibitory effects of odor substances, geosmin and 2-methylisoborneol, on early development of sea urchins. Wat. Res. 30:2508-2516.

NAQUADAT. 1976. National Water Quality Data Bank. Water Quality Branch. Inland Waters Directorate. Environment Canada, Ottawa, ON,.

NARTC [National Advances Resource Technology Center]. 1993. Aerial retardant application and use course. Feb. 16-19, 1993. Marana, AZ.

Nascimento, S.M. and S.M.F.O. Azevedo. 1999. Changes in cellular components in a cyanobacterium (*Synechocystis aquatilis* f. *salina*) subjected to different N/P ratios - an ecophysiological study. Environ. Toxicol. 14: 37-44.

NBARD [New Brunswick Agriculture and Rural Development]. 1997. Manure Management Guidelines for New Brunswick.

NBE [New Brunswick Environment]. 1996. Guidelines for issuing certificates of approval for the utilization of wastes as soil additives. 14 p.

Neill, C. 1990. Effects of nutrients and water levels on species composition in prairie whitetop (*Scholochloa festucacea*). Can. J. Bot. 68: 1015-1020.

Neilson. F.J.A. 1974. Nitrate/nitrite poisoning of stock. New. Zeal. J. Agr. 128: 43-45.

Newbold, J.D., J.W. Elwood, R.V. O'Neill and W. Van Winkle. 1981. Measuring nutrient spiralling in streams. Can. J. Fish. Aquat. Sci. 38: 860-863.

New Brunswick Environment. 1982. Effects of sewage treatment plant effluents on Phyllis Creek, Fredericton, New Brunswick, 1981. Environmental Services Branch, Fredericton, NB.

Nicholls, K. 1995. Chrysophyte blooms in the plankton and neuston of marine and freshwater systems. *In* Chrysophyte Algae. *Edited by* C. Sandgren, J. Smol,. and J. Kristiansen. Cambridge University Press, Cambridge. p. 181-213.

Nicholls, K.H. 1999. Effects of temperature and other factors on summer phosphorus in the inner Bay of Quinte, Lake Ontario: Implications for climate warming. J. Great Lakes Res. 25:250-262

Nicholls, K.H. and C.M. Cox 1978. Atmospheric nitrogen and phosphorus loading to Harp Lake, Ontario, Canada. Water Resour. Res. 14: 589-592.

Nicholls, K.H., D.W. Standen and G.J. Hopkins. 1980. Recent changes in the near-shore phytoplankton of Lake Erie's western basin at Kingsville, Ontario. J. Great Lakes Res. 6: 146-153.

Nicolson J.A. 1988. Alternate strip cutting in upland black spruce V: The impact of harvesting on the quality of water flowing from small basins in shallow-soil boreal ecosystems. For. Chron. 64: 52-58.

Nicolson, J.A., N.W. Foster and I.K. Morrison. 1982. Forest harvesting effects on water quality and nutrient status in the boreal forest. *In* Procedings of Canadian Hydrology Symposium 1982, Hydrological Processes of Forested Lands, National Research Council Canada. p. 71-89.

Nixon S.W., J.W. Ammerman, L.P. Atkinson, V.M. Berounsky, G.B. Billen, W.C. Boichourt, W.R. Boynton, T.M. Church, D.M. Di'Toro, R. Elmgren, J.H. Garber, A.E. Giblin, R.A. Jahnke, N.J.P. Owens, M.E.Q. Pilson and S.P. Seitzinger. 1996. The fate of nitrogen and phosphorus and the land-sea margin of the North Atlantic Ocean. Biogeochemistry 35: 141-180.

Norby, R.J., Y. Weerasuiya and P.J. Hanson. 1989. Induction of nitrate reductase activity in red spruce needles by NO_2 and HNO_3 vapour. Can. J. For. Res. 19: 889-896.

Nordin, R.N. 1985. Water Quality Criteria for Nutrients and Algae. Technical Appendix. Resource Quality Section, Water Management Branch, British Columbia Ministry of Environment, Victoria, BC.

Nordin, R.N. and L.W. Pommen. 1986. Water quality criteria for nitrogen (nitrate, nitrite, and ammonia). Ministry of Environment and Parks. Province of British Columbia. Water Quality Unit. Resource Quality Section. Water Management Branch. Victoria, BC. 11 p.

Northern Rivers Basin Study Board. 1996. Northern river basins study: Report to the Ministers, 1996. Alberta Environmental Protection - Corporate Management Service, Edmonton, AB. 287 p.

Norris, L.A., C.L. Hawkes, W.L. Webb, D.G Moore, W.B. Bollen and E. Holcombe. 1978. A report of research on the behaviour and impact of chemical fire retardants in forest streams. Forestry Science Laboratory. PNW Forest and Range Experiment Station.

NRC [National Research Council]. 1995. Nitarte and nitrite in drinking water. National Academy Press, Washington, DC.

NRC [National Research Council] Committee on Medical and Biological Effects of Environmental Pollutants. 1979. Ammonia, Division of medical sciences, Assembly of life science. University Park Press, Baltimore, MD.

NRC [National Research Council] Committee on Wastewater Management for Urban Coastal Areas. 1993a. Managing wastewater in coastal urban areas. National Academy of Sciences, Washington, DC, USA.

NRCan [Natural Resources Canada]. 1999. ARNEWS - Acid Rain National Early Warning System. <atl.cfs.nrcan.gc.ca/fhn/ma/arnews_e.htm> (6 Dec. 1999).

NSE [Nova Scotia Department of the Environment]. 1992. Nova Scotia standards and guidelines manual for the collection, treatment and disposal of sanitary sewage.

NWT [Northwest Territories] Water Board. 1992. Guidelines for the discharge of treated municipal wastewater in the Northwest Territories, Yellowknife, NWT.

NWWG [National Wetlands Working Group]. 1988. Wetlands of Canada. Ecological land classification series, No. 24. Sustainable Development Branch CWS. Conservation and protection. Environment Canada. 452 p.

Oberemm, A., J. Becker, G.A. Codd and C. Steinberg. 1999. Effects of cyanobacterial toxins and aqueous crude extracts of cyanobacteria on the development of fish and amphibians. Environ. Toxicol. 14: 77-88.

Odum, E.P. 1980. The status of three ecosystem-level hypotheses regarding salt marsh estuaries: Tidal subsidy, outselling, and detritus-based food chains. *In* Estuarine Perspective. *Edited by* V.S. Kennedy. Academic Press. New York. p. 485-495.

OECD [Organization for Economic Co-operation and Development]. 1995. OECD Environmental Data: Compendium 1995. Paris, France.

OECD [Organization for Economic Co-operation and Development]. 2000. Environmental Indicators for Agriculture Volume 3: Methods and Results. Paris, France.

Okano, K., T. Tptsila, T. Fukuzawa and T. Tazaki. 1985. Growth responses of plants of various concentrations of nitrogen dioxide. Environ. Pollut. 38: 361-373.

Oldham, R.S., D.M. Lathan, D Hilton-Brown, M. Towns, A.S. Cooke and A. Burn. 1997. The effect of ammonium nitrate fertilizer on frog (*Rana temporaria*) survival. Agric. Ecosys. Environ. 61(1): 69-74.

Olson, B.M., D.R. Bennett, R. Borg, R.H. McKenzie and R.P. Atkins. 1997a. Manure and nutrient management to sustain groundwater quality near feedlots in central Alberta. Canada-Alberta Environmentally Sustainable Agriculture Agreement, Lethbridge, AB.

Olson, B.M., R.H. McKenzie, T. Jensen, M.A. Arshad, P. Jedel, R.C. Izaurralde, D.W. McAndrew and A.M. Johnston. 1997b. Nitrate leaching in crop rotation studies in Alberta. Canada-Alberta Environmentally Sustainable Agriculture Agreement, Lethbridge, AB.

OMAFRA [Ontario Ministry of Agriculture, Food and Rural Affairs]. 1994. 1995-1996 Field crop recommendations. Queen's Printer for Ontario, Toronto, ON. Report RV-12-94-45M.

OMAFRA [Ontario Ministry of Agriculture, Food and Rural Affairs]. 1997. Best management practices: Nutrient management. Queen's Printer for Ontario, Toronto, ON. 69 p.

OME [Ontario Ministry of the Environment]. 1981. Outlines of analytical methods: a guide to the occurrence, significance, sampling and analysis of chemical and microbiological parameters in water, sediment, soil, vegetation, and air. Ontario Ministry of the Environment. Toronto, ON.

OME [Ontario Ministry of the Environment]. 1983. Rondeau Bay watershed data collection summary. M.O.E. Southwest Region, London, ON.

OME [Ontario Ministry of Environment]. 1984. Water management goals, policies, objectives and implementation procedures of the Ministry of the Environment. Revised. Toronto, ON. 70 p.

OME [Ontario Ministry of the Environment]. 1988. Thirty-seven municipal water pollution control plants: pilot monitoring study. Vol. 1, Interim Report and Vol. II, Appendix "A". Prepared by Canviro Consultants for the Water Resources Branch of the Ontario Ministry of the Environment.

OME [Ontario Ministry of Environment]. 1991. The physical and economic dimensions of municipal solid waste management in Ontario. Queen's Printer for Ontario, Toronto, ON.

OME [Ontario Ministry of Environment]. 1998. Rationale for a revised phosphorus criterion for precambrian shield lakes in Ontario. Standards Development Branch. Ontario Ministry of the Environment. Jan. 6 1998. Working Document. 17 p.

OMEE [Ontario Ministry of Environment and Energy]. 1993. Report on the 1991 discharge from municipal sewage treatment plants in Ontario. Vol. 1. Summary of performance and compliance. Toronto, ON.

OMEE [Ontario Ministry of Environment and Energy]. 1994. Water management policies, guidelines, and provincial water quality objectives. Toronto, ON. 31 p.

OMEE & OMAFRA [Ontario Ministry of Environment and Energy & Ontario Ministry of Agriculture and Rural Affairs]. 1996. Guidelines for the utilization of biosolids and other wastes on agricultural land. Toronto, ON.

Ontario Farm Environment Coalition. 1998. Ontario's Nutrient Management Strategy. <www.gov.on.ca/OMAFRA/english/environment/ofec/coalition.htm> (2 March 1998).

Oppenborn, J.B. and C.A. Goudie. 1993. Acute and sublethal effects of ammonia on striped bass and hybrid striped bass. J. World. Aqua. Soc. 24 (1): 90-101.

Østensvik, Ø. and O.M. Skulberg. 1981. Toxicity studies with blue-green algae from Norwegian inland waters. *In* The Water Environment: Algal Toxins and Health. *Edited by* W.W. Carmichael. Plenum Press, N.Y. p. 315-324.

Osweiler, G.D., T.L. Carson, W.B. Buck and G.A. Van Gelder. 1985. Clinical and diagnostic veterinary toxicology. 3rd ed. Kendall/Hunt Publishing Co. Dubuque, IA. p. 483.

Paerl, H.W. 1997. Coastal eutrophication and harmful algal blooms: importance of atmospheric deposition and groundwater as "new" nitrogen and other nutrient sources. Limnol. Oceanogr. 42: 1154-1165.

Paerl, H.W. and D.R. Whitall. 1999. Anthropogenically-derived atmospheric nitrogen deposition, marine eutrophication and harmful algal bloom expansion: Is there a link? Ambio 28:307-311

Painchaud, J. 1997. La qualité de l'eau des rivières du Québec: état et tendances. Ministère de l'Environnement et de la Faune, Direction des écosystèmes aquatiques, Québec. 58 p.

Painter, D.S., K.J. McCabe and W.L. Simser. 1989. Past and present limnological conditions in Cootes Paradise affecting aquatic vegetation. NWRI Contribution Paper # 88-47. Burlington, ON. 93 p.

Painter, D.S., L. Hampton and W.L. Simser. 1991. Cootes Paradise water turbidity: Sources and recommendations. NWRI Contribution Paper # 91-15. Burlington, ON. 18 p.

Palachek, R.M. and J.R. Tomasso. 1984. Toxicity of nitrite to channel catfish (*Ictalurus punctatus*), tilapia (*Tilapia aurea*) and largemouth bass (*Micropterus salmoides*): evidence for a nitrite exclusion mechanism. Can. J. Fish. Aquat. Sci. 41: 1739-1744.

Pang, P. C. K. and K. McCollough. 1982. Nutrient distribution in forest soil leachates after thinning and fertilizing Douglas-fir. Can. J. Soil Sci. 62: 197-208.

Paradis, D. 1997. Qualité de l'eau souterraine en zone de culture intensive de la pomme de terre dans la MRC de Portneuf. Ministère de l'Environnement et de la Faune, Québec. 14 p.

Paradis, D., P.J. Bernier and P. Levallois. 1991. Qualité de l'eau souterraine dans la MRC de Portneuf. Ministère de l'Environnement, Ministère de l'Agriculture, des Pêcheries et de l'Alimentation et Départment de santé communautaire du Centre Hospitalier de l'Université Laval, Québec. 13 p.

Patni, N.K. 1991. Overview of land appliation of animal manure in Canada. *In* Canadian Agricultural Research Council (CARC). *Edited by* D.A Leger, N.K Patni, S.K. Ho. Proceedings of the National Workshop on Land Application of Animal Manure. Ottawa, ON, June 1991. p. 7-17.

Paul, B.J. and H.C. Duthie. 1988. Nutrient cycling in the epilithon of running waters. Can. J. Bot. 67: 2302-2309.

PEI Agriculture. 2000. Guidelines for Manure Management for Prince Edward Island. <www2.gov.pe.ca/af/agweb/library/documents/manureguide/intro.asp> (16 Feb 2000).

Penczak, T., W. Galicka, M. Molinski, E. Kusto and M. Zalewski. 1982. The enrichment of a mesotrophic lake by carbon, phosphorus and nitrogen from the cage aquaculture of rainbow trout, *Salmo gairdneri*. J. Appl. Ecol. 19: 371-393.

Pérez-Soba, M., I. Stulen and L.J. Van der Eerden. 1990. Effects of NH_3 on the nitrogen metabolism of *Pinus sylvestris*. *In* Proceedings of the International Congress on Forest Decline Research,, Friedrichshafen, Germany, October. 1989. Karlsruhe, Germany, Centre for Nuclear Research (Poster summary).

Persson, G. 1991. Eutrophication resulting from salmonid fish culture in fresh and salt waters: Scandinavian experiences. *In* Nutritional Strategies and Aquaculture Waste. Proceedings of the First International Symposium on Nutritional Strategies in Management of Aquaculture Waste (NSMAW). *Edited by* C.B. Cowey and C.Y. Cho. University of Guelph, Guelph, ON, 1990.

Peters, R.H. 1973. Availability of atmospheric orthophosphate. J. Fish. Res. Bd. Can. 34: 918-924.

Peterson, B.J., L. Deegan, J. Helfrich, J.E. Hobbie, M. Hullar, M. Moller, T.E. Ford, A. Hershey, A. Hiltner, G. Kipphut, M.A. Lock, D.M. Fiebig, V. McKinley, M.C. Miller, J.R. Vestal, R. Ventullo and G. Volk. 1993. Biological responses of a tundra river to fertilization. Ecology. 74:653-672.

Peterson, B.J. and J.M. Melillo. 1985. The potential storage of carbon caused by eutrophication of the biosphere. Tellus 37b:117-127.

Petrie, B. and P. Yeats. 1990. Simple models of the circulation, dissolved metals, suspended solids and nutrients in Halifax Harbour. Water Pollut. Res. J. Can. 25: 325-349.

Petukhov, N.I. and A.V. Ivanov. 1970. Investigation of certain pyschophysiological reactions in children suffering from methemoglobinemia due to nitrates in water. Hyg. Sanit. 35: 29-41.

Phleger, F.B. 1971. Effect of salinity on growth of a salt marsh grass. Ecology 52: 908-911.

Pierce, R.H., J.M. Weeks and J.M. Prappas. 1993. Nitrate toxicity to five species of marine fish. J. World Aqua. Soc. 24 (1): 105-107.

Pierpont, R.A. and P.L. Minotti 1977. Effects of calcium carbonate on ammonium assimilation by tomato seedlings. J. Amer. Soc. Hortic. Sci. 102: 20-23.

Pilon, P. and M.A. Kerr. 1984. Land use monitoring on wetlands in the southwestern Fraser Lowland, British Columbia, Lands Directorate, Environment Canada. Working Paper No. 34. Vancouver, BC. 31 p. + maps.

Pinter, I.J. and L. Provasoli. 1963. Nutritional characteristics of some chrysomonads. *In* Symposium on Marine Microbiology. *Edited by* C.M. Oppehneimer. Thomas Publisher, Springfield, MA. p. 114-121.

Planas, D., M. Desrosiers, S-R. Groulx, S. Paquet and R. Carignan. 2000. Pelagic and benthic algal responses in eastern Canadian Boreal Shield lakes following harvesting and wildfires. Can. J. Fish. Aquat. Sci. 57 (Suppl. 2): 136-145.

Podbury, G. and C. Stushnoff. 2000. Risk of wind erosion. *In* Environmental sustainability of Canadian agriculture: Report of the Agri-Environmental Indicator Project. *Edited by* T. MacRae, C.A.S. Smith and L.J. Gregorich. Research Branch, Policy Branch, Prairie Farm Rehabilitation Administration, Agriculture and Agri-Food Canada.

Podemski, C.L. and J.M. Culp. 1996. Nutrient and contaminant effects of bleached kraft mill effluent on benthic algae and insects of the Athabasca River. *In* Environmental Fate and Effects of Pulp and Paper Mill Effluents. *Edited by* M.R. Servos, K.R. Munkittrick, J.H. Carey and G.J. Van der Kraak. St. Lucie Press, FL. p. 571-580.

Pomeroy, L.R. and R.G. Wiegert. 1981. The ecology of a salt marsh. Springer-Verlag. New York. 271 p.

Pommen, L.W. 1989. Approved and Working Criteria for Water Quality. Ministry of Environment. Water Management Branch. Resource Quality Section. Victoria, BC

Potash and Phosphate Insitute. 1988. Phosphorus for agriculture. Atlanta, GA.

Poulton, B., S. Hamilton, E. Hill, N. Vyas and D. Larson. 1993. Toxicity of fire retardants and foam suppressant chemicals to plants and animal communities: Progress Report. Interagency Fire Coordination Committee. Boise, ID.

Prairie, Y.T. 1994. Développement de modèles prédictifs décrivant l'effet de l'eutrophisation sur l'habitat du touladi (*Salvelinus namaycush*). Rapp. Min. Env. Faune. 33 p.

Prakash, A. and F.J.R. Taylor. 1966. A "Red water" bloom of *Gonyaulax acatenella* in the Strait of Georgia and its relation to Paralytic Shellfish Toxicity. J. Fish. Res. Bd. Can. 23: 1265-1270.

Preston, C.M., V.G. Marshall, G. McCollough and D.J. Mead. 1990. Fate of 15N-labelled fertilizer applied on snow at two sites in British Columbia. Can. J. For. Res. 20: 1583-1592.

Pries, J.H. 1994. Wastewater and Stormwater Applications of Wetlands in Canada. North American Wetlands Conservation Council (Canada). Ottawa, ON. Issues Paper No. 1994-1. 66 p.

Primeau, S. 1999. Le bassin de la rivière Yamaska : qualité des eaux de 1979 à 1997. *In* Le bassin de la rivière Yamaska : l'état de l'écosystème aquatique. Ministère de l'Environnement du Québec, QC.

Primeau, S., N. La Violette, J. St-Onge and D. Berryman. 1999. Le bassin de la rivière Yamaska: profil géographique, sources de pollution et interventions d'assainissement. *In* Le bassin de la rivière Yamaska: l'état de l'écosystème aquatique. Ministère de l'Environnement du Québec, QC.

Provasoli, L. and J.J.A. McLaughlin. 1963. Limited hetertrophy of some photosynthetic dinoflagellates. *In* Symposium on Marine Microbiology. *Edited by* C.M. Oppehneimer. Thomas Publisher, Springfield, MA. p. 105-113.

Ptacek, C.J. 1998. Geochemistry of a septic-system plume in a coastal barrier bar, Point Pelee, Ontario, Canada. J. Contam. Hydrol. 33: 293-312.

Ptacek, C.J., J. FitzGerald, A.S. Crowe, J.G. Bain, K.R. Waybrant and D.L. Thompson. 1998. Release of nutrients from on-site wastewater disposal systems, Point Pelee National Park, Ontario, Canada. *In* Proceedings of the Parks Research Forum of Ontario, 2nd Annual General Meeting, February 5-6, 1998, Peterborough, ON. p. 214-224.

Putnam, M.R. 1989. Toxicological problems in food animals affecting reproduction. Vet. Clin. N. Amer. Food Anim. Pract. 5: 325-344.

Ramade, F. 1987. Ecotoxicology. John Wiley and Sons Ltd. N.Y. 262 p.

Rasmussen, L., T. Brydges and P. Mathy. 1993. Experimental manipulations of biota and biogeochemical cycling in ecosystems. Report 4, Ecosystem Research Report Series. Commission of the European Communities, Brussels, Belgium. *Cited in* Vitousek, P.M., J.D. Aber, R.W. Howarth, G.E. Likens, P.A. Matson, D.W. Schindler, W.H. Schlesinger and D.G. Tilman. 1997. Human alteration of the global nitrogen cycle: sources and consequences. Ecol. Appl. 7: 737-750.

Raven, J.A. 1988. Acquisition of nitrogen by the shoots of land plants: Its occurrence and implications for acid-base regulation. New Phytol. 109: 1-20.

Redbo-Torsensson, P. 1994. The demographic consequence of nitrogen fertilization of a population of sundew, *Drosera rotundiferolia*. Acta. Bot. Neerl. 1994: 175-188.

Reddy, N.A. and N.R. Menon. 1979. Effects of ammonia and ammonium on tolerance and byssogensis in *Perna viridis*. Mar. Ecol. Prog. Ser. 1: 315-322.

Redner, B.D. and R.R. Stickney. 1979. Acclimatization to ammonia by *Tilapia aurea*. Trans. Amer. Fish. Soc. 108: 383-388.

Reesom, R., F.S. Soong, J. Fitzgerald, L. Turczynowicz, O. El Saadi, D. Roder, T. Maynard and I. Falconer. 1994. *Cited in* Health Canada 1998a. Health Effect of Toxic Cyanobacteria (Blue-Green Algae). National Health and Medical Research Council of Australia, Commonwealth Department of Human Services and Health. Australian Government Publishing Service, Canberra, Australia. 108 p. Report No. 98-06. 82 p.

Reynolds, W.D., C. A. Campbell, C. Chang, C. M. Cho, J. H. Ewanek, R.G. Kachanoski, J. A. MacLeod, P. H. Milburn, R. R. Simard. G. R. B. Webster and B. J. Zebarth. 1995. Chapter 10 Agrochemical entry into groundwater. *In* The health of our soils. *Edited by* D. F. Acton and L. J. Gregorich. Agriculture and Agri-Food Canada. Ottawa, ON.

Rice, S.D. and J.E. Bailey. 1980. Survival, size and emergence of pink salmon, (*Oncorhynchus gorbuscha*), alevins after short and long-term exposures to ammonia. Fish. Bull. 78: 641-648.

Rice, S.D. and R.M. Stokes. 1975. Acute toxicity of ammonia to several developmental stages of rainbow trout, *Salmo gairdneri*. U.S. Natl. Mar. Fish. Serv. Fish. Bull. 73: 207-211.

Ricklefs, R.E. 1990. Ecology, 3rd ed. W.H. Freeman, NY. 896 p.

Ridal J.J., B. Brownlee and D.R.S. Lean. 1999. Occurrence of odor compounds, 2-methylisoborneol and geosmin in eastern Lake Ontario and the upper St. Lawrence River. J. Great Lakes Res. 25: 198-204.

Ro, C., R. Vet, D. Ord and A Holloway. 1998. Canadian Air And Precipitation Monitoring Network (CAPMoN) Annual Summary Reports (1983-1996). Air Quality Reseach Branch, Atmospheric Environment Service, Environment Canada, Downsview, ON.

Robert, L. 1983. Caractérisation et variabilité des engrais de ferme. *In* Proceedings of Agri-Gestion Laval, Département d'économie rurale, Université Laval. December 8, 1993.

Robertson, H.E. and W.A. Riddell. 1949. Cyanosis of infants produced by high nitrate concentration in rural waters of Saskatchewan. Can. J. Pub. Hlth. 40: 72-77.

Robertson, W.D. 1995. Development of steady-state phosphate concentrations in septic system plumes. J. Contam. Hydrol. 19: 289-305.

Robertson, W.D. and J.A. Cherry. 1992. Persistence of nitrate in three septic system plumes on unconfined sands in Ontario. *In* Proceedings IAH Conference on Modern Trends in Hydrogeology, Hamilton, ON. May 10-14, 1992.

Robertson, W.D., J.A. Cherry and E.A. Sudicky. 1991. Ground water contamination from two small septic systems on sand aquifers. Ground water 29: 82-92.

Robertson, W.D., S.L. Schiff and C.J. Ptacek. 1998. Review of phosphate mobility and persistence in 10 septic system plumes. Ground water 36: 1000-1010.

Robinette, H.R. 1976. Effect of selected sublethal levels of ammonia on the growth of channel catfish (*Ictalurus punctatus*). Prog. Fish. Cult. 38: 26-29.

Robinson, B. and R. Gordon. 1996. Farm nutrient management expert system. Final report to: Canada/Nova Scotia Agreement on the Agricultural Component of the Green Plan. 21 p.

Robinson-Wilson, E.G. and W.K. Seim. 1975. The lethal and sublethal effects of a zirconium process effluent on juvenile salmonids. Water Res. Bull. 11: 975-986.

Rodgers, G.K.R., J. Vogt, D. Boyd, L. Simser, H. Lang, T. Murphy and D.S. Painter. 1992. Remedial Action Plan for Hamilton Harbour: Goals, Options and Recommendations. ISBN 0.7778-053302. Canada-Ontario Agreement Respecting Great Lakes Water Quality.

Rodvang, S. J., R. Schmidt-Bellach and L. I. Wassenaar. 1998. Nitrate in groundwater below irrigated fields in southern Alberta. Alberta Agriculture, Food and Rural Development, Lethbridge, AB.

Roelofs, J.M.G., A.J. Kempers, J.L.F.M. Houdijk and J. Jansen. 1985. The effect of air-borne ammonium sulphate on *Pinus nigra* var. maritime in the Netherlands. Plant Soil. 84: 45-56.

Roland, A.E. 1976. The coastal plain flora of Kejimkujik National Park. Unpublished report for Kejimkujik National Park, NS, Canada.

Rosage, T.F., R.M. Schutsky and K.M. Rapp. 1979. Toxicity of un-ionized ammonia to the spotfish shiner (*Notropic spilopterus*). Proc. Pa. Acad. Sci. 53: 39-42.

Rothenberg, S.J., A.R. Dahl, E.B. Barr and R.LK. Wolff. 1986. Generation, behaviour, and toxicity of ammonium sulphite aerosols. J. Air Pollut. Contr. Assoc. 36: 55-78.

Rotton, J., R.S. Tikofsky and H.T. Feldman. 1982. Behavioural effects of chemicals in drinking water. J. Appl. Psychol. 67: 230-245.

Rouse, J.D., C.A. Bishop and J. Struger. 1999. Nitrogen pollution: An assessment of its threat to amphibian survival. Environ. Hlth. Perspect. 107:799-803.

Rubin, A.J. and G.A. Elmaraghy. 1977. Studies on the toxicity of ammonia, nitrate, and their mixtures to guppy fry. Water Res. 11: 927-935.

Rudolph, D.L., D.A.J. Barry and M.J. Goss. 1998. Contamination in Ontario farmstead domestic wells and its association with agriculture: 2. Results from multilevel monitoring well installations. J. Contam. Hydrol. 32: 295-311.

Runnegar, M.T. and I.R. Falconer. 1981. Isolation, characterization and pathology of the toxin from the blue-green alga *Microcystis aeruginosa*. *In* The Water Environment: Algae Toxins and Health. *Edited by* W.W Carmichael. Plenum Press. N.Y. p. 325-342.

Russo, R.C. 1985. Ammonia, nitrite, and nitrate. *In* Fundamentals of Aquatic Toxicology. *Edited by* G. Rand and S.R. Petrocelli. Hemisphere Publishing Corporation. Washington. p. 455-471.

Russo, R.C., C.E. Smith and R.V. Thurston. 1974. Acute toxicity of nitrite to rainbow trout (*Salmo gairdneri*). J. Fish. Res. Bd. Can. 21: 1653-1655.

Ryding, S.-O. and W. Rast (eds.). 1989. The control of eutrophication of lakes and reservoirs. Volume 1. Man and the Biosphere Series. UNESCO, Parthenon Publishing Group Inc., Park Ridge, N.J.

Sabarathnam, S., G. Gupta and C. Mulchi. 1988. Nitrogen dioxide effects on photosynthesis in soyabean. J. Environ. Qual. 17: 143-146.

SAF [Saskatchewan Agriculture and Food]. 1997. Manual for Developing a Manure and Dead Animal Management plan.

Saiki, M.K., D.P. Moda. and B.L. Bellerud. 1999. Lethal levels of selected water quality variables to larval and juvenile Lost River and shortnose suckers. Environ. Pollut. 105 (1): 37-44.

Sandhu, R. and G. Gupta. 1989. Effects of nitrogen dioxide on growth and yield of black turtle bean (*Phaseolus vulgaris* L.) cv. Domino. Environ. Pollut. 59: 337-344.

Satchwill, T., S. Watson, E. Dixon and E. McCauley. 2000. Identification and treatment of off-flavours in drinking water. Proceedings, 9th CWWA National Conference on Drinking Water, Regina.

Savage. R.M. 1961. The ecology and life history of the common frog (*Rana temporaria temporaria*). Pitman and Sons Ltd. London, England. 236 p.

Scheider, W.A., W.R. Snyder and B. Clark. 1979. Deposition of nutrients and major ions by precipitation in south-central Ontario. Water Air Soil Pollut. 12: 171-185.

Schindler, D.W. 1974. Eutrophication and recovery in experimental lakes: Implications for lake management. Science 184: 897-899.

Schindler, D.W. and S.E. Bayley. 1993. The biosphere as an increasing sink for atmospheric carbon: Estimates from increasing nitrogen deposition. Global Biogeochem.l Cycles 7:717-734.

Schindler, D.W. and J.E. Nighswander. 1970. Nutrient supply and primary production in Clear Lake eastern Ontario. J. Fish. Res. Bd. Can. 27: 2009-2036.

Schindler, D.W., J. Kalff, H.E. Welch, G.J. Brunskill, H. Kling and N. Kritsch. 1974. Eutrophication in the high arctic. Meretta Lake, Cornwallis Island (75° N lat.). J. Fish. Res. Bd. Can. 31: 585-607.

Schindler, D.W., R.W. Newbury, K.F. Beaty and P. Campbell. 1976. Natural water and chemical budgets for a small Precambrian lake basin in central Canada. J. Fish. Res. Bd. Can. 33: 2526 - 2543.

Schleef, K.H. and W. Kleinhanss. 1994. Mineral Balances in Agriculture in the EU. Institute of Farm Economics, Federal Agricultural Research Centre, Braunschweig, Germany.

Schlesinger, R.B., L.C. Chen, I. Finkelstein and J.T. Zelikoff. 1990. Comparative potency of inhaled acidic sulfates: speciation and the role of hydrogen ion. Environ. Res. 52: 210-235.

Schmidt, C.W. 1997. Amphibian deformities continue to puzzle researchers. Environ. Sci. Tech. 31(7): 324- 326A.

Schramm, W. and P.H. Nienhuis. 1996. Marine Benthic Vegetation: Recent Changes and the Effects of Eutrophication. Springer-Verlag. Berlin. 470 p.

Schwimmer, M. and D. Schwimmer. 1968. Medical aspects of phycology. *In* Algae, Man, and the Environment. *Edited by* D.F. Jackson. Syracuse University Press, Syracuse NY. p. 279-359

Schulze, E.D. 1989. Air pollution and forest decline in a spruce (*Picea abies*) forest. Science 244:776-783.

Scrimgeour, G.J. and P.A. Chambers. 1996. Identification of Spatial Patterns in Nutrient Limitation with Herbivory Effects, Wapiti, Smoky and Athabasca Rivers, 1994. Prepared for the Northern River Basins Study, Edmonton, AB. Report No. 96.

Scrimgeour, G.J. and P.A. Chambers. 2000 Cumulative effects of pulp mill and municipal effluents on epilithic biomass and nutrient limitation in a large northern river ecosystem. Can. J. Fish. Aquat. Sci. 57: 1342-1354.

Scrivener, J.C. 1988. Changes in concentrations of dissolved ions during 16 years at Carnation Creek, British Columbia. *In* Proceedings from the Workshop: Applying 15 years of Carnation Creek resulpacific biological Station, Nanaimo, BC. Edited by T.W. Chamberlin. p. 75-80.

Scully, N.M., P.R. Leavitt and S.R. Carpenter. 2000. Century-long effects of forest harvest on the physical structure and autotrophic community of a small temperate lake. Can. J. Fish. Aquat. Sci. 57 (Suppl. 2): 50-59.

Semkin, R.G., A.W. McLarty and D. Craig. 1976. A water quality study of Cootes Paradise. Ontario Ministry of Environment, West Central Region, Toronto, ON.

SERM [Saskatchewan Environment and Resource Management]. 1983. Water quality objectives. Jan. 1975, 3rd. Printing. Water Pollution Control Branch, Regina, SK.

SERM [Saskatchewan Environment and Resource Management]. 1995. Surface water quality objectives. Environmental protection and sustainable resource management. MB# 110- reprinted. Regina, SK. 15 p.

SERM [Saskatchewan Environment and Resource Management]. 1998. Draft guide for land disposal of sewage. Saskatoon, SK.

Sessions, S.K., R.A. Franssen and V.L. Horner. 1999. Morphological clues from multilegged frogs: Are retinoids to blame? Science. 284: 800-802.

Shaw, R.D., A.M. Trimbee, A. Minty, H. Fricker and E.E. Prepas. 1989. Atmospheric deposition of phosphorus and nitrogen in central Alberta with emphasis on Narrow Lake. Water Air Soil Pollut. 43: 119-134.

Shelton, I.J., G.J. Wall, J.-M. Cossette, R. Eilers, B. Grant, D. King, G. Padbury, H. Rees, J. Tajek and L. van Vliet. 2000. Risk of water erosion. *In* Environmental sustainability of Canadian agriculture: Report of the agri-environmental indicator project. *Edited by* T. MacRae, C.A.S. Smith and [illegible] Gregorich. Research Branch, Policy Branch, Prairie Farm Rehabilitation Administration, Agriculture and Agri-Food Canada.

Sheppard, L.J., R.I. Smith and M.G.R. Cannell. 1989. Frost hardiness of *Picea rubens* growing in spruce declined regions of the Appalachians. Tree Physiol. 5: 25-37.

Shilo, M. and M. Shilo. 1953. Conditions which determine the efficiency of ammonia sulphate in the control of *Prymnesium parvum* in fish breeding ponds. Appl. Microbiol. 1: 330-333.

Shutter, S. B., E. A. Sudicky and W. D. Robertson. 1994. Chemical fate and transport in a domestic septic system: Application of a variably saturated model for chemical movement. Environ. Toxicol. Chem. 13: 223-231.

Shuval, H.I. and N. Gruener. 1972. Epidemiological and toxicological aspects of nitrates and nitrites in the environment. Amer. J. Pub. Hlth. 62: 1045-1062.

Sibert, J.R. 1982. Roles for Detritus in Complementing Productivity of Coastal Systems. *In* Synthetic and Degradation Processes in Marine Macrophytes. *Edited by* L.M. Srivastava. Walter de Gruyter, N.Y. p. 207-214.

Siegrist, R., M. Witt and W. Boyle. 1976. Characteristics of rural household wastewater. J. Environ. Eng. ASCA 102: 533-548.

Sims, R.A. 1949. Phytoplankton studies of Cootes Paradise Marsh, Royal Botanical Gardens, Hamilton MA Thesis. McMaster University, Hamilton, ON.

Singer, S.N. 1977. Preliminary estimates of urban storm water unit loadings for the Canadian Great Lakes Basin. Tech. Memorandum, Ministry of the Environment, Toronto, July.

Simard, A. and J. Painchaud. 2000. Qualité de l'eau des rivières du Québec, 1988-1998. Vecteur Environnement 33: 70-74.

Sjörs, H. 1950. On the relation between vegetation and electrolytes in north Swedish mire waters. Oikos 2: 241-258.

Skulberg, O.M., G.A. Codd and W.W. Carmichael. 1984. Toxic blue-green algal blooms in Europe: a growing problem. Ambio. 13: 244-247.

Smayda, T.J. 1997. Harmful algal blooms: their ecophysiology and general relevance to phytoplankton blooms in the sea. Limnol. Oceanogr. 42: 1137-1153.

Smith, C.A.S. and T. McRae. 2000. Understanding and assessing the environmental sustainability of agriculture. *In* Environmental sustainability of Canadian agriculture: Report of the Agri-Environmental Indicator Project. *Edited by* T. MacRae, C.A.S. Smith and L.J. Gregorich. Research Branch, Policy Branch, Prairie Farm Rehabilitation Administration, Agriculture and Agri-Food Canada.

Smith, D.C. 1987. Adult recruitment in chorus frogs: effects of size and date of metamorphosis. Ecology 68: 344-350.

Smith, G.J. 1987. Pesticide use and toxicology in relation to wildlife: Organophosphate and carbamate compounds. Fish and Wildlife Services, U.S. Dept. Int. Washington, DC. Resource Publication 170. 128 p.

Smith, J.C., R. Cormier, J. Worms, C.J. Bird, M.A. Quilliam, R. Pocklington, R. Angus and L. Hanic. 1990. Toxic blooms of the domoic acid containing diatom *Nitzschia pungens* in the Cardigan River, PEI, in 1988. *In* Toxic Marine Phytoplankton. *Edited by* E. Granéli, B. Sundström, L. Edler, D.M. Anderson. Proceedings of the Fourth International Conference on Toxic Marine Phytoplankton. June 26-30, 1989. Lund, Sweden. p. 227-232.

Søballe, D.M. and B.L. Kimmel. 1987. A large-scale comparison of factors influencing phytoplankton abundance in rivers, lakes and impoundments. Ecology 68: 1943-1954.

Somers, G. 1998. Distribution and trends for the occurrence of nitrate in P.E.I. groundwaters. Proceedings of Workshop on Nitrate - Agricultural Sources and Fate in the Environment - Perspectives and Direction, Charlottetown, PEI. February 26, 1998.

Sommerfeldt, T.G., U.J. Pittman and R.A. Milne. 1973. Effect of feedlot manure on soil and water quality. J. Environ. Qual: 1071-1078.

Sommerfeldt, T.G. and C. Chang. 1985. Changes in soil properties under annual applications of feedlot manure and different tillage practices. Soil Sci. Soc. Am. J. 49: 983-987.

Sonzogni, W.C., S.C. Chapra, D.E. Armstrong and T.J. Logan. 1982. Bioavailability of phosphorus inputs to lakes. J. Environ. Qual. 11: 555-563.

Sosiak, A.J. and D.O. Trew. 1996. Pine Lake Restoration Project: Diagnostic Study (1992). Surface Water Assessment Branch, Technical Services and Monitoring Division, Alberta Environmental Protection.

Sosiak, A.J. 2000. Long-term response of periphyton and macrophytes to reduced municipal nutrient loading to the Bow River (Alberta, Canada). Alberta Environment draft report.

Spehar, R.L., A.E. Lemke, Q.H. Pickering, T.H. Roush, R.C. Russo and J.D. Yount. 1981. Effects of pollution on freshwater fish. J. Water Pollut. Cont. Fed. 53: 1028-1076.

Speijers, G.L.A., G.F. Van Went, M.E. Van Apeldoorn, G.K. Montizaan, J.A. Janus, J.H. Canton, C.A.M. Van Gestel, C.A. Van der Heijden, E. Heijna-Merkus, A.G.A.D. Knaap, R. Luttik and D. De Zwart. 1989. Integrated Criteria Document Nitrate. Appendix to Report No. 758473012. National Institute of Public Health and Environmental Protection, Bilthoven, The Netherlands. 39 p.

Squires, L. and A.G. Vander Valk. 1992. Water depth tolerances of the dominant emergent macrophytes of the Delta Marsh, Manitoba. Can. J. Bot. 70: 1860-1867.

St. Lawrence Vision 2000. 1998. <www.slv2000.qc.ec.gc.ca/slv2000/english/library/doc_center/protection/list.htm> (November 1998).

Stanley Associates Engineering Ltd. 1992. Urban runoff quantification and contaminants loading in the Fraser River and Burrard Inlet. Prepared for the Fraser River Action Plan, Environment Canada. DOE FRAP Report No. 1993-19. 105 p.

Stanley, R.A. 1974. Toxicity of heavy metals and salts to Eurasian watermilfoil (*Myriophyllum spicatum*). Arch. Environ. Contam. Toxicol. 2(4): 331-341.

Statistics Canada. 1992. Household Facilities and Equipment. Catalogue No. 64-202. Ottawa, ON.

Statistics Canada. 1994. Human Activity and the Environment 1994. Statistics Canada Catalogue No. 11-528, Ottawa, ON.

Statistics Canada. 1997a. Agricultural Profile of Canada. Ottawa, ON.

Statistics Canada. 1997b. Historical Overview of Canadian Agriculture. Ottawa, ON.

Statistics Canada. 1997c. Household Facilities and Equipment. Catalogue No. 64-202. Ottawa, ON.

Statistics Canada. 1997d. Manufacturing Industries of Canada: national and provincial areas 1996. Catalogue No. 31-203-XPB. Ottawa, ON.

Statistics Canada. 1998a. Population. <www.statcan.ca/english/Pgdb/People/Population/demo02.htm> (2 Nov. 1998).

Statistics Canada. 1998b. Waste management industry survey: Business sector, 1995. Ottawa, ON.

Steedman, R.J. 2000. Effects of experimental clearcut logging on water quality in three small boreal forest lake trout (*Salvelinus namaycush*) lakes. Can. J. Fish. Aquat. Sci. 57 (Suppl. 2): 92-96.

Steen, I. 1998. Phosphorus availability in the 21st century: management of a non-renewable resource. Phosphorus & Potassium 217: 25–31. <www.nhm.ac.uk/mineralogy/phos/p&k217/steen.htm> (22 September 1999).

Steppuhn, H., E. Coxworth, J.A. Kernan, D.G. Green, J.E. Knipfel. 1994. Response of *Kochia scoparia* to nitrogen fertilization on a saline soil. Can. J. Soil. Sci. 74: 267-275.

Stewart A.R., G.A. Stern, W.L. Lockhart, A. Salki, M.P. Stainton, B.N. Billeck, R. Danell, J. Delaronde, N.P.Grift, T. Halldorson, K. Koczanski, A. MacHutcheon, B. Rosenburg, D. Savoie, D. Tenkula, G. Tomy and A. Yarachewski. 1999. The influence of the 1997 Red River flood on contaminant transport and fate in Southern Lake Winnipeg. A report to the International Joint Commission though the International Red River Basin Task Force.

Stewart, B.A. 1970. A look at agricultural practices in relation to nitrate accumulation. *In* Nutrient mobility in soils: accumulation and losses. *Edited by* O.P. Engelstad and M. Stelly. Special Publ. No. 4. Soil Sci. Soc. Amer. Inc., Madison WI.

Stillmeadow Laboratories. 1991. Summary of fire chemical toxicity tests: Fire-Trol LCA-R (raffinate). Data submitted to U.S. Forest survey. Sugarland, TX.

Stockner, J. and E. MacIsaac. 1996. British Columbia lake enrichment programme: Two decades of habitat enhancement for Sockeye salmon. Regulated Rivers: Res. Manage. 12: 547-561.

Stombaugh, D.P., H.S. Teague and W.L. Roller. 1969. Effects of atmospheric ammonia on the pig. J. Anim. Sci. 28: 844-856.

Stoner, R.D., W.H. Adams, D.N. Slatkin and H.W. Siegelman. 1989. The effects of single L-amino acid substitutions on the lethal potencies of microcystins. Toxicon. 27: 825-828.

Strain, P. 1998. Eutrophication capacity of Maritime inlets. *In* Proceedings of the Toxic Chemicals Progress and Directions Workshop. *Edited by* H. Vandermeulen. Department of Fisheries and Oceans. Hull, QC, October, 1998.

Strain, P.M., D.J. Wildish and P.A. Yeats. 1995. The application of simple models of nutrient loading and oxygen demand to the management of a marine tidal inlet. Mar. Pollut. Bull. 30: 253-261.

Switzer-Howse, K.D. and D.R. Coote. 1984. Agricultural practices and environmental conservation. Agriculture Canada Publication No. 1772/E. Ottawa, ON. 27 p.

Taylor, F.J.R., R. Haigh and T.F. Sutherland. 1994. Phytoplankton ecology of Sechelt Inlet, a fjord system on the British Columbia coast. II. Potentially harmful species. Mar. Ecol. Progr. Ser. 103: 151-164.

Taylor, L.A., P.M. Chapman, R.A. Miller and R.V. Pym. 1995. Victoria's Wastewater Discharges: Effects on the Marine Environment. CRD, 16 p.

Taylor, O.C. 1973. Oxidant Air Pollution Effects on a Western Coniferous Forest Ecosystem. Task C Report. University of California Statewide Air pollution research centre. Riverside CA, USA. 139 p pp.

Taylor, O.C. and F.M Eaton. 1966. Suppression of plant growth by NO_2. Plant Physiol. 41: 132-135.

Tchobanoglous, G. and F. Burton. 1991. Wastewater Engineering: Treatment, Disposal and Reuse/ Metcalf & Eddy, Inc., 3re ed. McGraw-Hill, New York. 1334 p.

Teal, J and M. Teal. 1969. Life and death of the salt marsh. Boston Little Brown, Boston, MA. 278 p.

Teshow, M. and F.K. Anderson. 1989. Plant stress from air pollution. John Wiley and Sons. New York. 283 p.

Thompson, D.R., G. Kats and E. Hensel. 1970. Effects of ambient levels of NO_2 on navel oranges. Environ. Sci. Tech. 5: 1017-1019.

Thomson, R.E. 1994. Physical oceanography of the Strait of Georgia-Puget Sound-Juan de Fuca Strait system. *In* Review of the Marine Environment and Biota of Strait of Georgia, Puget Sound and Juan de Fuca Strait: Proceedings of the BC/Washington Symposium on the Marine Environment, Jan. 13 & 14, 1994. *Edited by* R.C.H.Wilson, R.J. Beamish, F. Aitkens and J. Bell. Can. Tech. Rep. Fish. Aquat. Sci. No. 1948. p. 36-100.

Thorp, S., R. Rivers and V. Pebbles. 1997. Impacts of Changing Land Use. State of the Lakes Ecosystem Conference 1996 Background Paper. Environment Canada and United States Environmental Protection Agency. EPA 905-R-97-015d.

3Rs Information Partnership (Ontario Ministry of Environment, CSR: Corporations Supporting Recycling, Recycling Council of Ontario, Association of Municipal Recycling Coordinators and the Solid Waste Association of North America). 1999. Municipal 3Rs in Ontario: 1997 Factsheet. <www.rco.on.ca/factsheet/fs_f08.html > (22 June 1999).

Thurston, J.M. 1969. The effect of liming and fertilizers on the botanical composition of permanent grassland and the yield of hay. *In*. Ecological Aspects of the Mineral Nutrition of Plants. *Edited by* I.H. Rouson. Blackwell Scientific Publication. Oxford, UK.

Thurston, R.V., G.R. Phillips, R.C. Russo and S.M. Hinkins. 1981. Increased toxicity of ammonia to rainbow trout (*Salmo gairdneri*) resulting from reduced concentrations of dissolved oxygen. Can. J. Fish. Aquat. Sci. 38: 983-988.

Thurston, R.V., R.C. Russo and C.E. Smith. 1978. Acute toxicity of ammonia and nitrate to cutthroat trout fry. J. Amer. Fish. Soc. 107(2): 361-368.

Thurston, R.V.M.S., R.C. Russo and G. Phillips. 1983. Acute toxicity of ammonia to rainbow trout and fathead minnows. Trans. Amer. Fish. Soc. 112: 705-711.

Tilman, D. 1986. Biodiversity: population versus ecosystem stability. Ecology. 77: 350-363.

Tilman, D. 1987. Secondary succession and the pattern of plant dominance along experimental nitrogen gradients. Ecol. Monogr. 57: 189-214.

Tilman, D. and J.A. Downing. 1994. Biodiversity and stability in grasslands. Nature. 367:363-365.

Todd, B.J., C.F.M. Lewis, L.H. Thorleifson, and E. Neilson, *Eds.* 1996. Lake Winnipeg Project: cruise report and scientific results; Geological Survey of Canada Open File 3113, 656pp.

Todd, E.C.D. 1990. Amnesic shellfish poisoning- a new seafood toxin syndrome. *In* Toxic marine phytoplankton. *Edited by* E. Granéli, B. Sundström, L. Edler, D.M Anderson. Proceedings of the Fourth International Conference on Toxic Marine Phytoplankton. June 26-30, 1989. Lund, Sweden. p. 504-508.

Tomasso, J.R. and G.J. Carmichael. 1986. Acute toxicity of ammonia, nitrite, and nitrate to the Guadalupe Bass, *Micropterus treculi*. Buil. Environ. Contam. Toxicol. 36: 866-870.

Townsend, A.R., B.H. Braswell, E.A. Holland and J.E. Penner. 1996. Spatial and temporal patterns in terrestrial carbon strorage due to deposition of fossil fuel nitrogen. Ecol. Appl. 6: 806-814.

Trama, F.B. 1954. The acute toxicity of some salts of sodium, potassium and calcium to the common bluegill (*Lepomis marochirus* Rafinesque). Proc. Acad. Nat. Sci. Philadelphia 106: 185-205.

Transport Canada. 1972. Manual of snow removal and ice control. Ottawa, ON. Cited in Transport Canada 1990.

Transport Canada. 1990. The environmental impact of urea use on airport runways. Safety and technical services. Ottawa, ON. TP 10069E. 41 p. + Appendices.

Transport Canada. 1994. A comparison of the effects of urea, potassium acetate, calcium magnesium acetate, and sodium formate runway de-icers on the environment. Safety and Technical services. Ottawa, ON. TP 12285E. 49 p.

Tsujl, K., S. Nalto, F. Kondo, N. Ishikawa, M.F. Watanabe, M. Suzki and K-I Harada. 1994. Stability of microcystins form cyanobacteria: effect of light on decomposition and isomerization. Environ. Sci. Tech. 28: 173-177.

Turek, B., D. Hlavsova, J. Turek, J. Waldman and J. Cerna. 1980. The fate of nitrates and nitrites in the organisms. IARC Sci. Publ. 31: 625-641.

Ueno, Y., Y. Makita, S. Negata, T. Tsutsumi, F. Yoshida, S-I. Tamura, M. Sekijima, F. Tashiro, T. Harada and T. Yoshida. 1999. No chronic oral toxicity of a low dose of microcystin-LR, a cyanobacterial hepatotoxin, in female BALB/ c mice. Environ. Toxicol. 14: 45-55.

UK Environment Agency. 1998. Aquatic eutrophication in England and Wales. Environmental Issues Series, Environment Agency, Bristol, UK.

UMA Engineering Ltd. 1993. Sewer use control for Fraser River Basin and Burrard Inlet Drainage Basin. Environment Canada, Conservation and Protection, Vancouver, BC. Fraser River Action Plan Report. 84 p.

UMA Engineering Ltd. 1994. Inventory of Municipal Stormwater Discharges within the Fraser River Estuary. Environment Canada, Pacific and Yukon Region, Vancouver, BC. Fraser River Action Plan Report 1993-38.

UMA Engineering Ltd. 1995. Combined Sewer Overflow Inventory for the Fraser River Basin and Burrard Inlet. Environment Canada, Pacific and Yukon Region, Vancouver, BC. Fraser River Action Plan Report 1993-21.

United Nations Environment Programme (UNEP). 1995. Global Programme of Action for the Protection of the Marine Environment from Land-Based Activities. Intergovernmental Conference to Adopt a Global Programme of Action for the Protection of the Marine Environment from Land-Based Activities. Washington, DC, October 23 - November 3, 1995. UNEP(COA)/LBA/IG.2/7.

Urban, N.R. and S.E. Bayley. 1986. The acid-base balance of peatland: a short term perspective. Water Air Soil Pollut. 30: 791-800.

Urban, N.R. and S.J. Eisenreich. 1988. N cycling in a forested Minnesota Bog. Can. J. Bot. 66: 435-449.

US Census Bureau. 1998. US Census Bureau: The Official Statistics. Population Topics. <www.census.gov/ftp/pub/population/www/>.

USDA (United States Department of Agriculture). 1997. Agricultural Resources and Environmental Indicators, 1996-1997. Economic Research Service, Natural Resources and Environment Division, US Department of Agriculture, Agricultural Handbook No. 712, Washington, DC.

US Department of Agriculture. 1992. Agricultural waste waste mangement field handbook. National engineering handbook. Soil Conservation Service.

USEPA [U.S. Environmental Protection Agency]. 1974. Urban stormwater management and technology: An assessment. USEPA, Cincinnati, Ohio, Rept. EPA-670/2-74-040.

USEPA [U.S. Environmental Protection Agency]. 1983. Results of the nationwide urban runoff program: Volume I-final report. Water Planning Division, U.S. Environmental Protection Agency, NTIS PB84-185552.

USEPA [U.S. Environmental Protection Agency]. 1994a. National water quality inventory: 1994 report to Congress. EPA841-R-95-005. Washington, D.C.

USEPA [U.S. Environmental Protection Agency]. 1994b. National water quality inventory: 1994 report to Congress, appendixes. EPA841-R-95-006. Washington, D.C.

USEPA [U.S. Environmental Protection Agency]. 1998a. National strategy for the development of regional nutrient criteria. USEPA Rept. EPA 822-R98-002.

USEPA [United States Environmental Protection Agency]. 1998b Our nation's waters: Summary of the national water quality inventory. Report to Congress EPA 841-S00-001.

Usher, R. and J. Scarth. 1990. Alberta's wetlands: Water in the bank! ECA 90-PA/CS-514. Environmental Council of Alberta, Edmonton, AB. 37 p.

Vaithiyanathan, P., C.J. Richardson, G.R.G. Kavanaugh, C.R. Craft and T. Barkay. 1996. Relationship of eutrophication to the distribution of mercury and the potential for methyl mercury production in the peat soils of the Everglades. Environ. Sci. Tech. 30: 2591-2597.

Valiela, I. and J.M. Teal. 1979. The N budget of a salt marsh ecosystem. Nature 280: 653-656

Valiela, I., J.M. Teal and W.J. Sass. 1975. Production and dynamics of salt marsh vegetation and the effects of experimental treatment with sewage sludge: biomass, production, and species composition. J. Appl. Ecol. 12: 973-982.

Van der Eerden, L.J.M. 1982. Toxicity of ammonia to plants. Agr. Environ. 7: 223-235.

Van der Eerden, L.J.M., T.A. Duek, J.J.M. Berdowski, H. Greven and H.F. Van Dobben. 1991. Influence of NH_3 and $(NH_4)_2SO_4$ on heathland vegetation. Acta. Bot. Neerl. 40: 281-296.

Van Dijk, H.F.G. and J.G.M. Roelofs. 1988. Effects of excessive ammonium deposition on the nutritional status and condition of pine needles. Physiol Plant. 73: 494-501.

Van Haut, H. and B. Prinz. 1979. Evaluation of relative phytotoxicity of organic air pollutants in the LIS short-term test. Stau Reinhalt Luft. 39: 408-413.

Van Leeuwen, J.A. 1998. Human and animal health implications of elevated nitrate intake. *In* Proceedings of Workshop on Nitrate - Agricultural Sources and Fate in the Environment - Perspectives and Direction, Charlottetown, February 26, 1998.

Verhoeven, J.T.A. and M.B. Schmitz. 1991. Control of plant growth by nitrogen and phosphorus in mesotrophic fens. Biogeochem. 12: 135-148.

Vermeer, H.J.G. 1986. The effect of nutrients on shoot biomass and species composition of wetland and hayfield communities. Acta. Ecol. Plant. 7: 31-41.

Vermeer, J.G. and F. Berendse. 1983. The relationship between nutrient availability, shoot biomass and species richness in grassland and wetland communities. Vegetation. 53: 121-126.

Vézina, C. 1997. Ammonia Emission Inventory, 1995. Environment Canada. Unpublished data.

Vincent, P., G. Dubois and H. Leclerc. 1983. Nitrate in drinking water and cancer mortality. Epidemiological study in the north of France. Rev. Epidemiol. Santé Publique. 31: 199-214.

Vitt, I.J., J.P. Caldwell, H.M. Wilbur and D.C. Smith. 1990. Amphibians as harbingers of decay. Bioscience 40: 418.

Vitousek, P.M., J.D. Aber, R.W. Howarth, G.E. Likens, P.A. Matson, D.W. Schindler, W.H. Schlesinger and D.G. Tilman. 1997. Human alteration of the global nitrogen cycle: sources and consequences. Ecol. Appl. 7: 737-750.

Vitousek, P.M. and R.W. Howarth. 1991. Nitrogen limitation on land and in the sea: how can it occur? Biogeochem. 13: 87-115.

Vogel J., E. Royal and V. Snoeyink. 1997. Zebra mussels linked to taste, odor problems. American Water Works Association. Opflow 23: 1, 4-5.

Vogelsang, G.F. and D.B. Kent. 1997. Shallow groundwater survey summary report. Report prepared by Clifton Associates Ltd. for Saskatchewan Environment and Resource Management, Regina.

Vollenweider, R.A. 1976. Advances in defining critical loading levels for phosphorus in lake eutrophication. Mem. Ist. Ital. Idrobiol. 33:53-83.

Vollenweider, R.A. 1992: Coastal marine eutrophication: principles and control. *In* Marine Coastal Eutrophication: The Response of Marine Transitional Systems to Human Impact: Problems and Perspectives for Restoration, Proceedings of an International Conference, Bologna, Italy, 21-24 March 1990. *Edited by* R.A. Vollenweider, R. Marchetti and R. Viviani. Sci. Total Environ (1992 Suppl.): 1-20.

Vollenweider, R.A., A. Rinaldi and G. Montanari. 1992. Eutrophication, structure and dynamics of a marine coastal system: results of ten-year monitoring along the Emilia-Romagna coast (Northwest Adriatic Sea). *In* Marine Coastal Eutrophication: The Response of Marine Transitional Systems to Human Impact: Problems and Perspectives for Restoration, Proceedings of an International Conference, Bologna, Italy, 21-24 March 1990. *Edited by* R.A. Vollenweider, R. Marchetti and R. Viviani. Sci. Total Environ (1992 Suppl.): 63-106.

Vyas, N.B., J.W. Spann, W.N. Beyer and E.F. Hill. 1994. Toxicity of fire retardant chemicals to wildlife species. *In* Foam Applications for Wildland and Urban Fire Management. *Edited by* National Wildfire Coordinating Group and Canadian Committee on Forest Fire Management. International wildland fire foam symposium and workshop, Thunder Bay, ON. May 3-7, 1994. 24 p.

Waller, D.H. and W.C. Hart. 1986. Solids, nutrients, and chlorides in urban runoff. *In* Urban Runoff Pollution. *Edtied by* H.C. Torno, J. Marsalke and M. Desbordes. NATO ASI (Advanced Science Institute) Series G: Ecological Sciences Vol. 10. Springer-Verlag, Berlin. p. 59-85.

Ward, P. 1992. Wetlands of the Fraser River Lowlands, 1989: Summary Report. Technical report series No. 156. Canadian Wildlife Service, Pacific and Yukon Region, BC. 36 p.

Warman, P.R. 1997. Alternative amendments in agriculture: overview of their characteristics and use. *In* Alternative Amendments in Agriculture. Proceedings of the Joint Symposium on Alternative Amendments in Agriculture. *Edited by* P.R Warman. Agricultural Institute of Canada, Truro, NS, 1997.

Wassenaar, L.I. 1995. Evaluation of the origin and fate of nitrate in the Abbotsford Aquifer using the isotopes ^{15}N and ^{18}O in NO_3^-. Appl. Geochem.10: 391-405.

Watson, S. and E. McCauley. 1988. Contrasting patterns of net- and nannoplankton production and biomass among lakes. Can. J. Fish. Aquat. Sci. 45:915-920.

Watson, S., E. McCauley and J. Downing. 1997. Patterns in phytoplankton taxonomic composition across temperate lakes of differing nutrient status. Limnol. Oceanogr. 42: 486-495.

Watson, S., B. Brownlee, T. Satchwill, and E. McCauley 1999. The use of solid phase microextraction (SPME) to monitor for major organoleptic compounds produced by chrysophytes in surface waters. Wat. Sci. Technol. 40:251-256

Watson, S., B. Brownlee, T. Satchwill. and E. Hargesheimer. 2000. Quantitative analysis of trace levels of geosmin and MIB in source and drinking water using headspace SPME. Water. Res. 34:2818-2828

Watt, P.J. and R.S. Oldham. 1995. The effect of ammonium nitrate on the feeding and development of larva of the smooth newt, *Triturus vulgaris* (L), and on the behavior of its food source, *Daphnia*. Freshwat. Biol. 33 (2): 319-324.

Webber. M.D. 2000. Study 11 – Industrial organic compunds in selected Canadian municipal sludges and agricultural soils. <res.agr.ca/lond/pmrc/nscp/sqep.html#Study11> (January 31, 2000).

Webber, M.D. and T.E. Bates. 1997. Municipal sewage sludge use on agricultural land. *In* Alternative Amendments in Agriculture. Proceedings of the Joint Symposium on Alternative Amendments in Agriculture. *Edited by* P.R. Warman. Agricultural Institute of Canada, Truro, NS, 1997.

Webster, J.R. and B.C. Patten. 1979. Effects of watershed perturbation on stream potassium and calcium dynamics. Ecol. Monogr. 49: 51-72.

Wedeles, C.H.R., J.D. Meisner and M.J. Rose. 1992. Wetland science research needs in Canada. Wetland Research Centre. Waterloo, ON. 30 p.

Wedemeyer, G.A. and W.T. Yasatake. 1978. Prevention and treatment of nitrite toxicity in juvenile steelhead trout (*Salmo gairdneri*). J. Fish. Res. Bd. Can. 35: 822-827.

Wederkinch. E. 1988. Population size, migration barriers, and other feathers of *Rana dalmatina* populations near Koge, Zealand, Denmark. Memo. Soc. Fauna Flora Fenn. 64: 101-103.

Weetman, G.F., Krause, H.H., Koller, E. and Veilleux, J.M. 1987. Interprovincial forest fertilization trials 5- and 10-year results. For. Chron. 63: 184-192.

WEF [Water Environment Federation]. 1998. Biosolids: A Short Explanation and Discussion. WEF/U.S. EPA Biosolids Fact Sheet Project. Water Environment Web: <www.wef.org/docs/biofact/explanation.html>.

Weisner, S.E.B., J.A. Strand and H. Sandsten. 1997. Mechanisms regulating abundance of submerged vegetation in shallow eutrophic lakes. Oecologia 109: 592-599.

Weiss, G. 1986. Hazardous Chemicals Data Book, 2nd ed. Noyes Data Corporation, Park Ridge, New Jersey. 1069 p.

Welch, E.B. 1992. Ecological Effects of Wastewater: Applied Limnology and Pollutant Effects, 2nd ed. Chapman and Hall, London. 425 p.

Welch, H.E. and J.A. Legault. 1986. Precipitation and chemistry limnology of fertilized and natural lakes at Saqvaqjuac, Northwest Territories. Can. J. Fish. Aquat. Sci. 43: 1104-1134.

Wellburn, A.R., J. Wilson and P.H. Aldridge. 1980. Biochemical responses on nitric oxide polluted atmosphere. Environ. Pollut. A22: 219-228.

Wellburn, A.R., C. Higginson, D. Robinson and C. Walmsey. 1981. Biochemical explanation of more than additive inhibitory low atmospheric levels of SO_2 and NO_2 upon plants. New Phytol. 88: 223-237.

West, E.S., W.R. Todd, H.S. Mason and J.T. Van Bruggen. 1966. Textbook of biochemistry, 4th ed. MacMillan Co., New York.

West, P., T.M. Fyles, B. King and D.C. Peeler. 1994. The effects of human activity on the marine environment of the Georgia Bas*In* Present waste loadings and future trends. *In* Review of the Marine Environment and Biota of Strait of Georgia, Puget Sound and Juan de Fuca Strait: Proceedings of the BC/Washington Symposium on the Marine Environment, Jan. 13 & 14, 1994. *Edited by* R.C.H. Wilson, R.J. Beamish, F. Aitkens and J. Bell. Can. Tech. Rep. Fish. Aquat. Sci. No. 1948. p. 9-35.

Westin, D.T. 1974. Nitrate and nitrite toxicity to salmonid fishes. Prog. Fish. Cult. 36: 86-89.

Wetzel, R.G. 1983. Limnology, 2nd ed. Saunders College Publishing, Fort Worth, TX. 767 p.

Whelan, B. R. and Z. T. Titmanis. 1982. Daily chemical variability of domestic septic tank effluent. Water Air Soil Pollut. 17: 131-139.

Whillans, T. 1996. Historical and comparative perspectives on rehabilitation of marshes as habitat for fish in the lower Great Lakes basin. Can. J. Fish. Aquat. Sci. 53 (Suppl. 1): 56-66.

WHO [World Health Organization]. 1977. Nitrates, nitrites, and N-nitroso compounds. Environmental Health Criteria 5. Geneva. 107 p.

WHO [World Health Organization]. 1986. Ammonia. Environmental Health Criteria No. 54. World Health Organization, Geneva. 432 p.

WHO [World Health Organization]. 1997a. Nitrogen oxides, 2nd ed.. Environmental Health Criteria No. 188. World Health Organization, Geneva. 550 p.

WHO [World Health Organization]. 1997b. Report of the Working Group Meeting on Chemical Substances in Drinking Water. World Health Organization, Geneva. Cited in Falconer 1999.

Wicks, R.J. and P.J. Thiel. 1990. Environmental factors affecting the production of peptide toxins in floating scums of the cyanobacterium *Microcystis aeruginosa* in a hypereutrophic African Reservoir. Environ. Sci. Tech. 24: 1413-1418.

Wildish, D.J., V. Zitko, H.M. Akagi and A.J. Wilson. 1990. Sedimentary anoxia caused by salmonid mariculture wastes in the Bay of Fundy and its effects on dissolved oxygen in seawater. *In* Proceedings from the Canada-Norway Finfish Aquaculture workshop, Sept. 11-14, 1989. St. Andrews, NB Can. Tech. Rep. Fish. Aquat. Sci. No. 1761. p. 11-18.

Wildish, D.J., P.D. Keizer, A.J. Wilson and J.L. Martin. 1993. Seasonal changes of dissolved oxygen and plant nutrients in seawater near salmonid net pens in the macrotidal Bay of Fundy. Can. J. Fish. Aquat. Sci. 50: 303-311.

Wilhelm, S.R., S.L. Schiff and W.D. Robertson. 1994a. Chemical fate and transport in a domestic septic system: unsaturated and saturated zone geochemistry. Environ. Toxicol. Chem. 13: 193-203.

Wilhelm, S.R., S.L. Schiff and J.A. Cherry. 1994b. Biogeochemical evolution of domestic wastewater in septic systems, 1. Conceptual model. Ground Water 32: 905-916.

Williams, M.C. 1989. Accumulation of nitrate by annual goldeneye and showy goldeneye. J. Range Mgmt. 42(3): 196-198.

Williamson, D.A. 1983. Surface water quality management proposal. Vol. 1. Surface Water Quality Objectives. Water Standards and Studies Report No, 83-2. Manitoba Department of Environment and Workplace Safety and Health, Winnipeg, MB.

Williamson, D.A. 2000. Manitoba Water Quality Standards, Objectives, and Guidelines. Water Quality Management Section. Manitoba Conservation Winnipeg, MB. . 75 p. Draft April 20. 2000.

Willis, A.J. 1963. Bruanton Burrows: the effects of the vegetation of the addition of mineral nutrients to dune soils. J. Ecol. 51: 353-374.

Wilson, E.J. 1992. Foliar uptake and release of inorganic nitrogen compounds in *Pinus sylvestris L.* and *Picea abis (L) Karst*. New Phytol. 120: 407-416.

Wilson, S.D. and P.A. Keddy. 1988. Species richness, survivorship, and biomass accumulation along an environmental gradient. Oikos. 53: 375-380.

Wingsle, G., T. Näsholm, T. Lundmark and A. Ericsson. 1987. Induction of nitrate reductase in needles of Scots pine seedlings by NO_x and NO_3. Physiol. Plant. 70: 399-403.

Winnipeg Web. 1999. Combined Sewers and River Quality <www.mbnet.mb.ca/city/html/services/browater/sewers.htm> (24 March 1999).

Winterbourn, M.J. 1990. Interactions among nutrients, algae and invertebrates in a New Zealand mountain stream. Freshwat. Biol. 23: 463-474.

Wisheu, I.C. and P.A. Keddy. 1994. The low competitive ability of Canada's Atlantic coastal plain shoreline flora: Implications for conservation. Biol. Conser. 68: 247-252.

Wisheu, I.C., P.A. Keddy, D.R.J. Moore, S.J. McCanny and C.L. Gaudet. 1991. Effects of eutrophication on wetland vegetation. *In* Proceedings of an International Symposium. Wetlands of the Great Lakes: Protection and Restoration Policies; Status of the Science. *Edited by* J. Kusler and R. Smardon. Niagara Fails, N.Y. May 16-18, 1990. Managers, Inc. N.Y. p. 112-121.

Wohlgemuth, K. and H.H. Casper. 1992. Nitrate poisoning of livestock. Nrth Dakota State University. Fango, North Dakota. U.S. Web page:<iaia.ext.nodak.edu/extpubs/ansci/livestoc/v839w.htm.>

Wollenweber, B. and J.A. Raven. 1993. Implications of nitrogen acquisition from atmospheric ammonia for acid-base and cation-anion balance in *Lolium perenne*. Physiol. Plant. 89: 519-523.

Wright, J.L.C., R.K. Boyd, A.S.W. De Freitas, M. Falk, R.A. Foxall, W.D. Jamieson,. M.V. Laycock, A.W. McColloch, A.G. McInnes, P. Odense, V. Pathak, M.A. Quilliam, M.A. Ragan, P.G. Sim, P. Thibault J.A. Walter, M. Gilgan, D.J.A. Richard and D. Dewar. 1989. Identification of domoic acid, a neuroexcitatory amino acid, in toxic mussels from eastern Prince Edward Island. Can. J. Chem. 67: 481-490.

Wrona, F.J., W.D. Gummer, K.J. Cash and K. Crutchfield. 1996. Cumulative impacts within the northern river basins. Northern River Basins Study, Environment Canada and Alberta Environmental Protection, Edmonton, AB. Synthesis Report No. 11.

Wyman, R.L. 1990. What's happening to the amphibians?. Conser. Biol. 4(4): 350-352.

Xu, Q. and R.S. Oldham. 1997. Lethal and sublethal effects of nitrogen fertilizer ammonium nitrate on a common toad (*Bufo bufo*) tadpoles. Arch. Environ. Contam. Toxicol. 32: 298-303.

Yang, Y.S., J.M. Skelly and B.I. Chevonne. 1983. Effects of pollutant combinations at low doses on growth of forest trees. Aquilo. Ser. Bot. 19: 406-418.

Yangdong, P. and R.L. Lowe. 1995. The effects of hydropsychid colonization on algal response to nutrient enrichment in a small Michigan stream, U.S.A. Freshwat. Biol. 33: 393-400.

Yaremcio, B. 1991. Nitrate poisoning and feeding nitrate feeds to livestock. Government Report # 08. Alberta Livestock Branch. Edmonton, AB. 7 p.

Zebarth, B.J., B. Hii, H. Liebscher, K. Chipperfield, J.W. Paul, G. Grove and S.Y. Szeto. 1998. Agricultural land use practices and nitrate contamination in the Abbotsford Aquifer, British Columbia, Canada. Agr. Ecosys. Environ. 69: 99-112.

Zebarth. B.J., J.W. Paul and K. Chipperfield. 1999. Nutrient losses to a soil from field storage of solid poultry manure. Can. J. Soil Sci. 79: 183-189.

Appendices

Appendix 1: Nitrogen and phosphorus applied as fertilizer in Canada 1981-1996. N and P data from Agriculture and Agri-Food Canada 1998. Area of application data from Statistics Canada 19997a.

Province	**Year**	**Area of Application (ha)**	**N Applied (t)**	**P Applied (t)**
Newfoundland	1981	4,414		
	1986	4,757		
	1991	5,434		
	1996	6,464		
PEI	1981	107,442		
	1986	113,297		
	1991	102,117		
	1996	119,541		
Nova Scotia	1981	88,537		
	1986	85,043		
	1991	82,267		
	1996	88,552		
New Brunswick	1981	75,997		
	1986	84,048		
	1991	78,136		
	1996	90,740		
Atlantic	1981	276,390	19,000	9,159
	1986	287,145	21,157	8,694
	1991	267,954	23,336	9,177
	1996	305,297	26,229	9,909
Quebec	1981	1,105,401	61,400	26,230
	1986	1,189,233	89,443	33,175
	1991	996,722	91,760	28,263
	1996	991,062	88,207	25,302
Ontario	1981	2,533,823	205,600	65,833
	1986	2,591,126	213,916	58,432
	1991	2,273,448	174,860	43,913
	1996	2,407,516	173,884	43,132
Manitoba	1981	3,196,455	154,448	39,526
	1986	3,726,374	235,960	45,334
	1991	3,688,335	259,890	44,554
	1996	3,830,391	312,350	51,838
Saskatchewan	1981	5,525,854	167,734	60,223
	1986	8,125,031	317,800	79,781
	1991	7,654,551	247,346	57,491
	1996	10,015,619	517,829	93,058
Alberta	1981	5,505,172	302,154	66,488
	1986	6,854,875	308,908	66,478
	1991	6,349,884	337,094	60,277
	1996	7,031,430	430,782	69,327
Prairies	1981	14,227,481	624,336	166,237
	1986	18,706,280	862,668	191,593
	1991	17,692,770	844,330	162,322
	1996	20,877,440	1,260,961	214,223
British Columbia	1981	362,105	27,485	5,654
	1986	374,186	33,537	7,003
	1991	330,937	23,477	4,950
	1996	361,320	26,922	4,706
Canada	1981	18,505,200	937,821	273,113
	1986	23,147,970	1,220,721	298,897
	1991	21,561,831	1,157,763	248,626
	1996	24,942,635	1,576,203	297,272

Appendix 2: Manure nitrogen and phosphorus production coefficients per animal type by province.

		NF[1]	PEI[1]	NS[2]	NB[3]	QC[4]	ON[5]	MB[6]	SK[7]	AB[8]	BC[9]
N	(kg/animal/yr)										
Dairy	Bulls	-	-	-	-	32.1	-	-	93.1	106.5	112.0
	Cows	88.7	88.7	88.7	98.4	55.7	64	98.6	93.1	106.5	116.0
	Heifers	40.7	40.7	40.7	-	21.4	-	46.4	69.8	32.0	42.0
	Steers	-	-	-	-	8.6	-	11.7	21.0	2.2	20.0
	Calves	-	-	-	-	-	-	-	-	-	1.7
Beef	Bulls	-	-	-	-	23.6	-	58.4	58.4	23.2	112.0
	Cows	65.0	65.0	65.0	67.0	32.1	32	58.4	58.4	23.2	78.0
	Heifers	40.1	40.1	40.1	44.6	19.3	-	55.9	43.4	11.8	44.0
	Steers	-	-	-	-	21.4	-	55.8	55.8	11.8	50.0
	Calves	-	-	-	-	7.5	-	31.0	15.7	4.6	20.0
Swine	Boars	3.3	3.3	3.3	11.4	5.5	11	12.8	12.0	16.4	24.3
	Sows	14.6	14.6	14.6	14.6	14.8	16	21.6	12.0	22	18.3
Poultry	Chickens	0.31	0.31	0.31	0.60	0.98	0.35	0.53	0.60	0.30	0.60
	Turkeys	0.75	0.75	0.75	0.77	2.70	-	0.75	0.82	0.43	0.86
	Layers	0.51	0.51	0.51	0.51	1.49	0.53	0.73	-	0.98	0.80
	Pullets	0.15	0.15	0.15	0.15	-	-	0.24	-	0.34	0.34
P	(kg/animal/yr)										
Dairy	Bulls	-	-	-	-	9.0	-	-	16.0	21.3	20.1
	Cows	13.1	13.1	13.1	13.1	15.7	12.9	21.1	16.0	21.3	13.1
	Heifers	5.3	5.3	5.3	5.3	6.0	-	7.1	12.0	6.4	47.2
	Steers	-	-	-	-	2.4	-	2.4	3.6	0.4	21.9
	Calves	-	-	-	-	-	-	-	-	-	1.0
Beef	Bulls	-	-	-	-	6.6	-	19.5	19.5	6.6	20.1
	Cows	21.7	21.7	21.7	21.7	9.0	6.5	19.5	19.5	6.6	13.5
	Heifers	13.4	13.4	13.4	13.4	5.4	-	14.8	11.5	3.3	14.4
	Steers	-	-	-	-	6.0	-	14.8	14.8	3.3	16.2
	Calves	-	-	-	-	2.1	-	8.2	4.1	1.3	21.9
Swine	Boars	1.1	1.1	1.1	1.1	1.7	2.6	4.2	4.1	4.2	7.5
	Sows	4.8	4.8	4.8	4.8	5.4	3.9	7.1	4.1	6.0	5.6
Poultry	Chickens	0.10	0.10	0.10	0.10	0.19	0.07	0.15	0.07	0.09	0.23
	Turkeys	0.28	0.28	0.28	0.28	0.51	-	0.24	0.11	0.14	0.27
	Layers	0.19	0.19	0.19	0.19	0.28	0.18	0.17	-	0.44	0.23
	Pullets	0.06	0.06	0.06	0.06	-	-	0.06	-	0.09	0.10

[1] USDA (1992)
[2] Brenton and Mellish (1996)
[3] NB Agriculture and Rural Development (1997)
[4] Robert (1993)
[5] Fraser (1985)
[6] Manitoba Agriculture (1994)
[7] Saskatchewan Agriculture and Food (1997)
[8] AAFRD (2000)
[9] Brisbin (1995)

Appendix 3: Nitrogen and phosphorus applied as manure in Canada 1991-1996. N and P data calculated by multiplying the number of animals in each province (Statistics Canada 1997b) by the nitrogen and phosphorus content of manure for each province (from Appendix 2). Area of application data from Statistics Canada 1997a.

Province	Year	Area of Application (ha)	N Applied (t)	P Applied (t)
Newfoundland	1991	3,258	754	283
	1996	4,316	707	266
PEI	1991	26,028	3,542	1,172
	1996	32,453	3,484	1,130
Nova Scotia	1991	39,786	5,309	1,827
	1996	45,871	5,183	1,785
New Brunswick	1991	29,881	5,111	1,321
	1996	34,804	5,073	1,288
Atlantic	1991	98,953	14,716	4,602
	1996	117,444	14,446	4,469
Québec	1991	537,806	70,726	27,879
	1996	672,313	76,513	29,717
Ontario	1991	645,463	100,806	33,866
	1996	734,088	99,032	33,133
Manitoba	1991	162,337	63,096	15,015
	1996	208,796	45,468	18,450
Saskatchewan	1991	244,548	35,900	12,983
	1996	285,816	42,561	15,158
Alberta	1991	359,841	47,304	22,879
	1996	475,390	80,175	22,680
Prairies	1991	766,726	194,553	63,702
	1996	970,002	234,742	71,875
British Columbia	1991	78,208	23,866	15,106
	1996	85,133	25,612	15,696
Canada	1991	2,127,156	404,667	145,155
	1996	2,578,980	450,345	154,890

Appendix 4. Atmospheric N fixation rates for legumes and the amount of N removed with their harvest in Canada. Data from F. Selles and R. Lemke, Agriculture and Agri-Food Canada, personal communication.

Legume	**N fixed (kg/ha/d)**	**N removed Eastern Canada (kg N/ t crop)**	**N removed Western Canada (kg N/ t crop)**
Alfalfa			24
Soybeans	70	65	
Dry Beans	30	42	
Chickpeas	52.5		
Clover (various)	100		24
Sweet Clover	67		24
Fababean	80		37
Lentils	56		44
Peas	90		37
Legume Haylage	90	26	
Peas, Green		25	
Beans, Green		25	

Glossary[1]

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[1] Except where otherwise noted, terms are from or modified from Government of Canada 1996. Terms in italic are defined elsewhere in the Glossary.

A

acute toxicity: severe biological harm or death produced in an organism by a substance withina short time after exposure

algal bloom: See *bloom*.

ammonia nitrogen: Nitrogen in the form of ammonia (NH_3). See also *nitrogen fixation*.

anoxia: The absence of oxygen, which is necessary to sustain most life. In *aquatic ecosystems*, this refers to the absence of dissolved oxygen in water.

aquatic: Pertains to both freshwater and marine ecosystems.

B

benthic: Of or living on or in the bottom of a water body. See also *benthos*.

benthos: Organisms living on the bottom of bodies of water.

biochemical oxygen demand (BOD): A measure of the quantity of oxygen used in the biochemical oxidation of compounds containing carbon and nitrogen in a specified time, at a specified temperature, and under specific conditions. The standard measurement is made for five days at 20°C.

biologically available: The physical or chemical form of a substance that can be directly used by an organism.

bloom (also known as **algal bloom**): Rapid growth of certain algal constituents of *plankton* in and on a body of water that is so heavy as to colour the water. The rapid growth can be fuelled by enrichment of nutrients, such as phosphorus and nitrogen.

BOD: See *biochemical oxygen demand*.

C

chronic toxicity: a measurable, but not totally disabling, effect produced in an organism by a substance over a relatively long period. The end result of chronic toxicity can be death, although the usual effects are sublethal (e.g., inhibition of reproduction or growth).

coliform bacteria: A group of bacteria predominantly inhabiting the intestinal tracts of humans and other warm-blooded animals, but also occasionally found elsewhere. The total coliform group is commonly used as an *indicator* of the sanitary quality of water, because ingestion of these bacteria in drinking water can result in diseases such as cholera.

contamination: Introduction of any undesirable foreign substance — physical, chemical, or biological — into an ecosystem. Does not imply an effect (see *pollution*). Usually refers to the introduction of human-made substances.

cumulative effect: The effect on the environment that results from the incremental impact of an action when added to other past, present, and reasonably foreseeable future actions.

D

drainage basin: See *watershed*.

E

effective concentration [median ~] (EC_{50}) The concentration of a stressor estimated to be effective in producing a defined biological response, other than mortality, in 50% of the test organisms over a specific time interval (e.g., a 48-h *Daphnia* EC_{50} for immobilization).

effluent: A liquid waste material that is a by-product of human activity (e.g., municipal wastewater or liquid industrial waste), which may be discharged into bodies of water.

environmental indicator: A selected key statistic that represents or summarizes a significant aspect of the state of the environment, natural resource sustainability or related human activity. Environmental indicators focus on trends in environmental changes, the stresses that are causing them, how ecosystems and their components are responding to these changes, and societal responses to prevent, reduce or ameliorate these stresses. See also *indicator*.

eutrophic: Pertaining to a body of fresh water that is rich in nutrients. See also *trophic*, *trophic level*, and *trophic status*.

eutrophication (also known as **nutrient enrichment**): The process of over fertilization of a body of water by nutrients that produce more organic matter than the self-purification reactions can overcome. Eutrophication can be a natural process or it can be accelerated by an increase of nutrient loading to a water body by human activity. See also *trophic*, *trophic level*, and *trophic status*.

F

food chain: A food relationship in an ecosystem in which energy and nutrients are transferred through a series of organisms by each stage feeding on the preceding one and providing food for the succeeding stage. Each stage of a food chain is known as a trophic level. The first trophic level consists of the green plants that can undertake photosynthesis, thereby obtaining their energy from the sun. See also *food web*.

food web: A complex intermeshing of individual *food chains* in an ecosystem.

G

ground water: Water occurring below the ground surface, which may supply water to wells and springs. Ground water occupies pores, cavities, cracks, and other spaces in bedrock and unconsolidated surface materials.

H

habitat: The place or type of site where plant, animal, or microorganism populations normally occur. The concept of habitat includes the particular characteristics of that place, such as climate and the availability of water and other life requisites (e.g., soil nutrients for plants and suitable food and shelter for animals), which make it especially well suited to meet the life cycle needs of the particular wildlife.

hydrological cycle (also known as the **water cycle**): The complement of processes by which water reaches the Earth from atmospheric precipitation, passes through transport and storage stages on the Earth's surface, and is returned to the atmosphere through evaporation and evapotranspiration

I

indicator: A statistic or parameter measure that, tracked over time, provides information on trends in the condition of a phenomenon and has significance extending beyond that associated with the properties of the statistic itself. See also *environmental indicator.*

indicator species: An organism whose presence or absence suggests that certain ecological conditions prevail. An indicator species can be used to monitor how much of a factor is present and how an ecosystem is responding to stresses and changes.

intensive agriculture: large-scale agricultural production carried out on a relatively small land base.

L

leaching: Washing out of soluble substances by water passing down through soil. Leaching occurs when more water falls on the soil than is lost by evaporation from the surface. Rainwater running through the soil dissolves mineral nutrients and other substances and carries them via ground water into water bodies.

lethal concentration [median ~] (LC_{50}) The concentration of a stressor estimated to be lethal to 50% of the test organisms over a specific time interval (e.g., 96-h LC_{50})

loading: Total mass of contaminants to a water body or to the land surface over a specified time (e.g., tonnes per year of phosphorus).

M

macronutrient: An element, such as potassium or nitrogen, essential in large quantities for plant growth.

monitoring: The process of checking, observing, or keeping track of something for a specified period of time or at specified intervals

N

nitrogen fixation: Conversion of gaseous (atmospheric) nitrogen (N_2) to compounds such as ammonia (NH_3). Carried out in ecosystems mainly by certain bacteria and blue-green algae. See also ammonia nitrogen.

nitrogen oxides (NOx): A group of gases released by fossil fuel combustion, forest fires, lightning, and decaying vegetation. Nitrogen dioxide (NO_2), a reddish-brown gas with an irritating odour, is one of the key ingredients in smog. Nitrous oxide (N_2O) is a greenhouse gas whose principal source is agricultural soil in a degraded state.

no-observed-effect concentration (NOEC): The highest dose or concentration used in a toxicity test resulting in no statistically significant observed effect in the exposed organisms compared with control organisms.

non-point source: Source of pollution in which pollutants are discharged over a widespread area or from a number of small inputs rather than from distinct, identifiable sources. Examples include eroding croplands, urban and suburban lands, and logged forestlands. See also *point source*.

nutrient: Any element or compound that an organism must take in from its environment because it cannot produce it or cannot produce it as fast as it needs it. As pollutants, any substance or group of substances (e.g., phosphorus or nitrogen) that, if added to water in sufficient quantities, provides nourishment that promotes the growth of aquatic vegetation in those waters to such densities as to degrade or alter or form part of a process of degradation or alteration of the quality of those waters to an extent that is detrimental to their use by any plant or animal, including humans. An example would be eutrophication of a lake.

nutrient enrichment: See *eutrophication*.

O

organic compounds: Compounds based on carbon and usually also containing hydrogen, with or without oxygen, nitrogen, or other elements. Organic originally meant "of plant or animal origin," and it is still sometimes used in this way. For example, manure and sewage contain organic compounds of animal and plant origin. However, now that organic compounds are routinely created by people, the word "organic" is also used to refer to synthetic organic compounds, such as synthetic fertilizers.

organic matter: Plant, animal, or micro-organism matter, either living or dead.

oxygen depletion: Consumption of oxygen at a rate that exceeds the supply, usually from air. In a river or pond this may occur at night when oxygen production by plants ceases but oxygen consumption by plants and animals continues. In lakes that are sufficiently deep, oxygen depletion occurs naturally in the cold lower layer (hypolimnion) during the summer while it has no access to oxygen from the air.

P

pH: A numerical expression of the concentration of hydrogen ions in solution: pH 0–6.9 is acidic, pH 7 is neutral, and pH 7.1–14 is basic or alkaline.

phytoplankton: Microscopic aquatic vegetative life; plant portion of the plankton; the plant community in marine and freshwater situations that floats free in the water and contains many species of algae.

plankton: Collective noun for organisms that drift around in water because they are not capable of swimming against currents in the water. See also *phytoplankton* and *zooplankton*.

point source: A source of *pollution* that is distinct and identifiable. Includes outfall pipes from municipal sewage treatment plants and industrial plants. See also *non-point source*.

pollution: The release by humans, directly or indirectly, of substances or energy into ecosystems that results or is likely to result in such deleterious effects so as to harm living resources and life, be hazardous to human health, hinder human activities, impair the quality of the ecological resources, and reduce amenities.

primary wastewater treatment: First step in sewage treatment to remove large solid objects by screens (filters) and sediment and organic matter in settling chambers. See also *secondary wastewater treatment* and *tertiary wastewater treatment*.

S

secondary wastewater treatment: After *primary wastewater treatment*, removal of biodegradable organic matter from sewage using bacteria and other micro-organisms, inactivated sludge, or trickle filters. Also removes some of the phosphorus (30%) and nitrate (50%). See also *tertiary wastewater treatment*.

sublethal: involving a stimulus below the level that causes death.

T

tertiary wastewater treatment: Removal of nitrates, phosphates, organochlorine compounds, salts, acids, metals, and toxic organic compounds *after secondary wastewater treatment*. See also *primary wastewater treatment*.

toxic: Pertains to any substance if it is entering or may enter the environment in a quantity or concentration or under conditions having or that may have an immediate or long-term effect on the environment (including living organisms within it) or constituting or that may constitute a danger to human life or health.

toxicity: The inherent potential or capacity of a material to cause adverse effects in a living organism. See acute toxicity, chronic toxicity and sublethal.

trophic: Relating to processes of energy and nutrient transfer from one or more organisms to others in an ecosystem. See also *eutrophic*, *trophic level*, and *trophic status*.

trophic level: Functional classification of organisms in a community according to feeding relationships; the first trophic level includes green plants, the second level includes herbivores,and the third trophic level includes carnivores. See also *eutrophic*, *trophic*, and *trophic status*.

trophic status: A measure of the biological productivity in a body of water. Aquatic ecosystems are characterized as oligotrophic (low productivity), mesotrophic (medium productivity), or *eutrophic* (high productivity). See also *trophic* and *trophic level*.

turbid: Refers to water that is cloudy or murky as a result of suspended sediment. Water may become turbid as a result of soil erosion, from addition of effluents containing particulate matter, or through the churning up of bottom sediments (e.g., via boat traffic in a body of water).

turbidity: The state of being *turbid*.

W

watershed: An area of land that drains naturally into a stream or other waterway.

Z

zooplankton: Microscopic animals, usually invertebrates, that are found in the water of aquatic ecosystems. See also *phytoplankton* and *plankton*.